AIDS FOR THE CONSERVATION OF GREAT LAKES COASTAL MARSHES

AIDS FOR THE CONSERVATION OF GREAT LAKES COASTAL MARSHES: DEVELOPMENT OF A MACROPHYTE INDEX AND A NOVEL MACROPHYTE SAMPLING PROTOCOL

By
MELANIE VANESSA CROFT, B.SC

A Thesis

Submitted to the School of Graduate Studies
In Partial Fulfillment of the Requirements
For the Degree
Master of Science

McMaster University

MASTER OF SCIENCE (Biology) | McMaster University Hamilton, Ontario

TITLE: Aids for the conservation of Great Lakes Coastal marshes: the development of a macrophyte index and novel sampling protocol

AUTHOR: Melanie Vanessa Croft, B.Sc (University of Waterloo)

SUPERVISOR: Professor Patricia Chow-Fraser

NUMBER OF PAGES: xxii, 87

General Abstract

Wetlands are a valuable resource, providing many ecosystem services, but unfortunately, coastal wetlands in the Great lakes are under threat from human development, including water quality impairment, introduction of exotic invasive species as well as physical damage such as dredging, draining, and filling in of wetland habitat. These actions have a negative impact on the native flora and fauna in wetlands, making wetland conservation and important topic.

Wetland macrophytes play a vital role within wetlands, not only providing food for water fowl, migratory birds, fish, and mammals, but also providing the physical structure that is necessary for fish spawning, and they provide habitat for macroinvertebrates and zoobenthos. Different macrophyte species have been found to be associated with varying water quality conditions, and because of this wetland macrophytes are useful indicators of water quality conditions. I have developed a Wetland Macrophyte Index (WMI) using 127 wetlands throughout all five Great Lakes (Chapter 1), which relates plant species presence/absence data to water quality conditions, making it a useful indicator of fish habitat. The WMI was validated using historical data from two wetlands from before and after a remedial action plan was put in place and also it was successfully applied to two Canadian National Parks.

Information on the presence/absence of wetland macrophytes can be a very important tool in wetland conservation, but, unfortunately, there is no standard method for sampling macrophytes. In the second chapter I will compare two common macrophyte sampling methods (grid and transect) to a novel method (stratified) in six wetlands (three pristine and three degraded). The stratified method has proven to be beneficial for determining the macrophyte biodiversity within a wetland because more species, more unique species, and more rare species were found with the stratified method compared to the grid and transect methods.

Acknowledgements

Firstly, I would like to thank Pat Chow-Fraser for all her help and support during all phases of this project. None of this would have been possible without her advice, direction and good humour.

Thanks to Susan Dudley and Jonathan Stone for being on my supervisory committee and to two anonymous reviewers of Chapter 1.

I would also like to thank my fellow graduate students, Maja Cvetkovic, Kristina Kostuk, Sheila McNair, Titus Seilheimer, Lyndsay Smith and Anhua Wei for their help in the field and advice during data analysis. A special thanks to Maja for enduring many long days collecting plant data, in both the blazing heat and torrential downpours.

Thanks to my family for all their support during the last two years, and a very special thanks to Jeremy for all his love and encouragement.

Table of Contents

General Abstract iii
Acknowledgements iv
Table of Contents v
List of Tables vii
List of Figures ix
List of Appendices xi
General Introduction xii
  Canadian Wetlands xii
  Coastal Wetlands of the Laurentian Great Lakes xii
  The role of Wetlands xiii
  Wetland Conservation xiii
  The role of macrophytes within wetlands xiv
  Thesis Objectives xv
References xvi
Preface xvii
Chapter 1: Use and development of the Wetland Macrophyte Index to detect water-quality impairment in fish habitat of Great Lakes coastal marshes 1
  Abstract 2
  Introduction 3
  Methods 5
    Description of Study Sites 5
    Description of Overall Field Monitoring Programs 6
    Field Sampling for Development of the WMI 6
    Aquatic Plant Surveys 7
  Results and Discussion 9
    Development of the WMI 9
    U and T values for macrophyte taxa 11
    Comparison of WMI scores 13
    Accounting for Presence of Exotic Species 14
    Relationship Between WQI and the WMI 14
    Validation of the WMI: Cootes Paradise Marsh, Hamilton Harbour RAP 15
    Validation of the WMI: Sturgeon Bay, Severn Sound RAP 16
    Use of the WMI in Routine Monitoring Programs 18
  General Discussion 19
  Acknowledgments 24
  References 25
Chapter 2: Non-random sampling and its role in habitat conservation: A comparison of three wetland macrophyte sampling protocols 50
  Abstract 51
  Introduction 52
    Study Sites 54
  Methods 55
    Wetland surveys 55

Wetland Macrophyte Index (WMI) scores 56
Geographic data 56
Randomized re-sampling 57
Statistical analysis 57
Results 57
Species richness 58
Unique species 59
Uncommon species 59
Randomized re-sampling 59
Wetland Macrophyte Index (WMI) 60
Discussion 60
Conclusions 63
Acknowledgements 64
References 65

List of Tables

Table 1.1: Summary of environmental variables for the 154 wetland used for the CCA, where Turb = Turbidity, ISS = Inorganic Suspended Solids, OSS = Organic Suspended Solids, Chl a = Chlorophyll, Cond = Conductivity, TP = Total Phosphorus, TAN = Total Ammonia Nitrogen, TNN = Total Nitrate Nitrogen. Total Plant taxa refers to all emergent, submergent and floating plants, and # exotic taxa refers to only floating and submergent taxa. See text for explanation of WQI and the WMI scores. 30

Table 1.2: Summary of U and T values for all taxa included in this study, organized according to habit type (emergent, floating and submergent). Common names and species codes are also included for convenience. U value indicates the tolerance of a species to degradation (1= very tolerant, 5= very intolerant) and T value indicates the niche breadth (1=broad niche, 3= narrow niche). % occurrence indicates the percentage of wetlands (n=176) in which the species in question occurred. * denotes that the species is not native to North America. Some species may be found in more than one group (e.g. emergent and floating) depending on the season. 31

Table 1.3: Range of WMI scores associated with the different categories 34

Table 1.4: Summary of percent composition of submersed aquatic plant species found at 8 stations in Sturgeon Bay in 1988 and 2004. No significant difference between Shannon-Weiner diversity index for 1988 and 2004 (t-test, P= 0.52) 35

Table 1.5: Summary of the WMI scores associated with eight sites sampled for plants in Sturgeon Bay during 1988 and 2004. Distance from shore and proximity to sewage outflow pipe were determined by GIS (ArcView 3.2). Rank of proximity to sewage outflow pipe was given a value of 1 to 8 where the sampling site closest to the sewage outflow was given a value of 1 and the furthest was given a value of 8. Mean WMI scores from 1988 were significantly lower than 2004 (P<0.0021). 36

Table 1.6: Location and description of human impacts on wetlands of the two Canadian National parks. 37

Table 2.1: Total number of quadrats sampled with each method in the six study sites. 67

Table 2.2: Summary of total number of submergent, emergent, wet meadow and floating species recovered in each wetland by the three methods (GR=grid; ST=stratified; TR=transect), and when data from all three methods were combined (COMB). Numbers in bracket correspond to the percentage of species identified by each method relative to the total number of species recovered by all methods combined. 68

Table 2.3: Summary of 2-way ANOVA testing the effect of sampling method, wetland quality and interaction between sampling method and wetland quality on species richness of macrophytes in wetlands. 69

Table 2.4: Summary of 3-way ANOVA testing the effect of sampling method, wetland quality, plant group, and all possible interactions among these factors on species richness of macrophytes in wetlands. 70

Table 2.5: Comparison of the number of species unique to a single method on a site-by-site basis. 71
Table 2.6: Number of unique species grouped by plant groups: submergent (SUB), emergent (EM), floating (FL), and wet meadow (WM). 72
Table 2.7: Summary of 3-way ANOVA testing the effect of sampling method, wetland quality, plant group, and all possible interactions among these factors on the number of unique species in wetlands. 73
Table 2.8: Summary of statistics associated with regression analysis relating species richness to total quadrats y= species richness, x = total quadrats 74
Table 2.9: Summary of statistics associated with regression analysis relating species richness to corrected time (min) y= species richness, x = corrected time 75
Table 2.10: Number of quadrats required by each sampling method to survey 80% of the total species richness for each site, and the associated time required to conduct surveys. Values calculated using equations in Table 2.9 and Table 2.10. Average values at the bottom of the table are for the number of quadrats excluding data for the GR and TR from Cootes Paradise; numbers in brackets are averages calculated without excluding Cootes data. 76
Table 2.11: Comparison of WMI and WMIadj scores (calculated according to Croft and Chow-Fraser 2007) for data obtained from the three sampling methods in each wetland in this study. 77

List of Figures

Figure 1.1: Location of 176 wetland years used in the application of the WMI. Location of the four study sites used for validation of the WMI are indicated by square symbols. FFNMP=Fathom Five National Marine Park, ST=Sturgeon Bay, CP=Cootes Paradise, and PPNP=Point Pelee National Park. 38

Figure 1.2: Bi-plot of CCA Axis 1 versus CCA Axis 2. Vectors for the 11 environmental variables are shown (lines with arrows emanating from the origin). The strength of the correlation of the environmental variable with the axis is a direct function of the length of the vector and how close it is to the axis. Points represent the 154 wetland years used in the CCA. Wetlands were grouped after the CCA by lake, for ease of interpretation. Wetlands on the right side of the plot have higher nutrient and turbidity levels 39

Figure 1.3: Bi-plot of CCA Axis 1 versus CCA Axis 2. Vectors for the 11 environmental variables are shown (lines with arrows emanating from the origin). Some species centroids are identified with a 4-letter code (see Table 2 for explanation of codes). Species were grouped by habitat. Species found on the left of the plot are intolerant of degradation 40

Figure 1.4: Relationship between the WMI score and WQI score for 176 wetland-years a) wetlands grouped by lake, and b) open circles represent 154 wetland-years used for both the development of the WMI and for application of the WMI, closed circles represent the 22 wetland-years that were used only for the application of the WMI 41

Figure 1.5: a) Plot of the WMI (closed squares) and the WMIadj (open squares) vs. WQI score for 176 wetland-years. b) Plot of the WMI vs. WQI score corresponding to Cootes Paradise Marsh (open squares), and wetlands of Point Pelee National Park (closed circles) and Fathom Five National Marine Park (open circles). The regression lines correspond directly to those in Figure 1.5a. Dashed lines represent the 95% confidence intervals. 42

Figure 1.6: Relationship between the WMI (solid symbol) and the WMIadj (open symbol) for Cootes Paradise Marsh from 1946 to 2003. The WMI and WMIadj scores during the 1990s had the same value 43

Figure 1.7: Change in TP concentrations in Sturgeon Bay from 1972 to 2003. There was a decline in nutrient concentrations through time as indicated by a linear regression analysis (P=0.06). 44

Figure 1.8: Relationship between the WMI scores calculated for 8 stations in Sturgeon Bay between 1988 (closed circles), prior to RAP delisting and 2004 (open circles), following RAP delisting. There were significant differences between the two time periods (Paired t-test; P = 0.0012) 45

Figure 2.1: Map of the Laurentian Great Lakes showing the location of the six study sites, three in Eastern Georgian Bay, and three in Western Lake Ontario 78

Figure 2.2: Comparison of mean species richness (± SE) associated with three sampling methods for a) all six sites in this study and b) when sites are grouped according to environmental quality (open = pristine sites; solid = degraded sites) 79

Figure 2.3: Comparison of mean species richness (+ SE) associated with four plant groups sampled in pristine (open bars) and degraded (solid bars) sites.................. 80
Figure 2.4: Comparison of mean species richness (+ SE) associated with three sampling methods for pristine (open bars) and degraded (solid bars) sites. Data are presented separately for a) submergent taxa b) emergent taxa c) floating taxa and d) wet meadow taxa. 81
Figure 2.5: Cumulative number of species identified by each method for a) Black Rock b) Coffin Rock c) Thunder Bay d) Bronte Creek e) Cootes Paradise Marsh and f) Jordan Harbour. Horizontal lines indicate 100% (solid line) and 80% of species (dashed line) identified by all three methods. See Table 2.8 for corresponding r^2-values of method-specific regressions. 82
Figure 2.6: Amount of time required to complete macrophyte surveys using the three sampling methods for a) Black Rock, b) Coffin Rock, c) Thunder Bay, d) Bronte Creek, e) Cootes Paradise Marsh and f) Jordan Harbour. The time taken to identify species that were unique to only one method was subtracted from the total time. See Table 2.9 for corresponding r^2-values of method-specific regressions.. 83

List of Appendices

Appendix 1.1: List of 154 wetland years and their locations (latitude and longitude) used for the development of the WMI. 46

Appendix 2.1: Summary of depth measurements (cm) for species found in all 6 wetlands. Species have been divided into groups according to morphology. N indicates the number quadrats in which species were found. 84

General Introduction

Wetlands are described as areas with saturated or poorly drained soils with anoxic conditions and they contain species that have adapted to those conditions. There are five major classes of wetlands in Canada; bogs, fens, swamps, marshes and shallow open water (National Wetlands Working Group 1988). Bogs are peat accumulating systems that have no significant inflows or outflows (hydrology maintained by precipitation), and they support sphagnum mosses that have adapted to the low nutrient and acid conditions found within bogs. Fens are also considered to be peat accumulating systems but they receive drainage from the surrounding soil. They have higher nutrient content, are more minerotrophic and less acidic than bogs. Swamps are wetlands dominated by trees and shrubs and are characterized by continual or seasonal standing or gently flowing water. Marshes are considered to be rich in nutrients because of the periodic inundation by slow moving or standing water, which defines the characteristic growth of marsh vegetation such as reeds, rushes and sedges. Shallow open water wetlands lie at the transition between marshes and open water and are characterized by submergent and floating vegetation.

Canadian Wetlands

Canada is comparatively rich in terms of wetlands, with almost a quarter of the worlds wetlands lying within our borders (Cox 1993). It has been estimated that 14% (127 million hectares) of Canada's landmass is covered by wetland (National Wetlands Working Group 1988). The majority of the wetlands (88 %) found within Canada are peatland (bog or fen), with much of the peatland centered around Hudson Bay, in Ontario, Manitoba and the Northwest Territories. Ontario has the largest wetland area of all the Provinces and Territories in Canada (29 million hectares), and 77% of that is peatland.

Coastal Wetlands of the Laurentian Great Lakes

Wetlands account for 33% of Ontario's land area (29 million hectares) and 25% of Ontario's land area is peat land; that means only 8% (5 million hectares) of Ontario's land area is designated swamp, marsh or shallow open water wetland, and of that only a small percentage would be classified as coastal wetlands (National Wetlands Working Group 1988). Great Lakes coastal wetlands are defined as wetlands within two kilometers of one of the Great Lakes or it's connecting channels (EC and OMNR 2003), but for our study we only considered a wetland to be coastal if it was hydrologically connected to the Great Lakes or one of it's connecting channels. Many different kinds of coastal wetlands can be found on the Great Lakes, and there are several different wetland classification schemes that are used on both the Canadian and US sides of the border (Chow-Fraser and Albert 1999). The classification system developed for the Great Lakes Coastal Wetland Consortium (Albert et al. 2003) divides coastal wetlands into three main groups (lacustrine, riverine and barrier-protected), and from there, the wetlands are further subdivided according to geomorphic characteristics.

Only 236 coastal wetlands (53 619 hectares) have been evaluated on the Canadian Great Lakes (only wetlands greater than 2 ha were considered for evaluation), and it has been estimated that a further 173 wetlands have yet to be evaluated (EC and OMNR 2003). The majority of the unevaluated wetlands are in Lake Huron (97), specifically in Georgian Bay. Although coastal wetlands in the Great Lakes account for only a small proportion of the total wetland area in Canada they are essential habitat for many organisms. Great Lakes Coastal wetlands provide habitat for 80% of the roughly 200 fish species within the Great Lakes (Chow-Fraser and Albert 1999). Many species of migratory birds use coastal wetlands as staging grounds and provide them with a vital food source that enables them to continue their migration. There are over 40 species of birds that are dependent on Great Lakes coastal marshes including; geese, dabbling ducks, diving ducks, swans, warblers, gulls, and rails (National Wetlands Working Group 1988). Understanding the different types of wetlands and the different habitats they provide plays an important role in their conservation. Considering that 60-80% of the coastal wetlands within the Great Lakes have already been lost (Smith *et al.* 1991, Ball *et al.* 2003), it is imperative that appropriate conservation strategies are adopted before more wetlands are destroyed. The anthropogenic impacts to coastal wetlands in the Great Lakes ranges considerably from lake to lake, with the more pristine, oligotrophic wetlands located on the shores of Lake Superior, Lake Huron, and Georgian Bay; whereas the coastal wetlands in Lake Ontario and Erie are generally more degraded or eutrophic.

The role of Wetlands

Wetlands have been called "nature's kidneys" for their ability to filter waste and pollutants, as well as "nature's supermarket" for the high productivity and ability to provide a source of food to many organisms (Mitsch and Gosselink 2000). They are considered to be one of the most productive ecosystems on Earth. Wetlands provide many ecosystem services such as climate regulation, water storage, nutrient cycling, and biomass storage, as well as providing physical habitat and a source of food for birds, mammals, fish, amphibians, and reptiles. These ecosystems services are often considered in anthropocentric terms, such as money and lives saved because of their ability to control floods, storm damage, erosion, as well as water cleansing and contaminant removal (Cox 1993).

Wetlands are situated at the ecotone between the terrestrial and aquatic ecosystems. They play an important role in the biogeochemical cycling of nutrients, where they can act as both a source of nutrients, as well as a nutrient sink depending on conditions.

Wetland Conservation

Throughout history, wetlands have generally been held in poor opinion. During medieval times, wetlands were greatly feared and were thought to be the source of pestilence and disease, and to even harbour monsters and mythical creatures. Over the centuries, wetlands have been drained, dredged, and filled in to allow for agriculture, industrial and residential development as well as marine navigation. Wetlands were often considered to be wastelands, obstacles to progress, and something to be eradicated. Many wetlands were originally drained for agriculture, because wetland soils are rich in organic matter and were considered valuable for farming. The wetlands that have been fortunate enough to escape human intervention are generally found in remote areas, or locations that are somehow undesirable for development.

It has only been within the latter part of the 20th century that the intrinsic value of wetlands has become more widely acknowledged. Many wetlands in Canada have suffered similar fates as wetlands worldwide. During the last two and a half centuries we have lost and alarming proportion of our wetlands, with a loss of up to 98% in highly urbanized or agricultural areas (Cox 1993). Canadian government agencies have developed strategies such as the Great Lakes Wetlands Conservation Action Plan (GLWCAP) and the Canada-Ontario Agreement (COA) to address wetland conservation. Many of these strategies involve bringing together resources and expertise from different jurisdictions to improve monitoring and increase research. GLWCAP is specifically focused on the conservation and protection of Great Lake coastal wetlands. One of the long term goals of their strategic plan is to protect 30 000 hectares of existing coastal wetland in the Great Lakes by 2020. But, before a wetland can be conserved or protected, it must be assessed to determine its quality. This is generally done by characterizing the hydrology and geology of the wetland, followed by understanding the biota found there. The Ontario government has developed a wetland evaluation system (OMNR 1994) to determine wetland quality, but it is very labour intensive and largely focused on the upland portion of the wetland, with minimal emphasis on the aquatic portion related to fish habitat. Since the Fisheries Act is one of Canada's strongest pieces of environmental legislation it important that fish habitat be easily recognized, so that it can be protected.

The role of macrophytes within wetlands

Many of the earlier described ecosystem services that wetlands provide are accomplished by the macrophytes (aquatic plants). Macrophytes provide the physical structure within wetlands that offer protection for macroinvertebrates, zooplankton, and juvenile fish (Carpenter and Lodge 1986). Macrophytes release oxygen into the water from leaves and stems, and through their roots into the sediment, which can then be used by other organisms. They greatly influence nutrient chemistry within the water by acting as nutrient sinks, pulling nutrients from the water column but then acting as a source of nutrients to the sediment (Cronk and Fennessy 2001). Since macrophytes are primary producers they are one of the cornerstones in wetland food webs.

Macrophytes are also efficient at removing contaminants from the water, which has more recently led to the successful implementation of treatment wetlands. Macrophytes stabilize the shoreline sediment with their root structure, which helps prevent erosion. Macrophytes can ameliorate the effects of floods by slowing water flow, and by acting like a sponge, they can trap water in the accumulated organic matter. Macrophyte growth depends on light, water colour, currents and wave action, nutrient availability (N, P, C and micronutrients), temperature, number of growing days in the season and water chemistry (pH, alkalinity etc) (Wetzel 1988), and there are different species that have adapted to these varying conditions. Since there is such a diversity of macrophyte species that have adapted to different water quality conditions, ranging from degraded systems with high levels of nutrients and turbidity to pristine systems with low nutrients and turbidity, they are ideal organisms for use as indicators of wetland quality.

Thesis Objectives

Since many wetland functions rely, in some part, on the macrophytes, it is important that inventories and studies of wetlands consider species composition and distribution of the macrophytes. Currently within the Great Lakes, there is a push to develop indicators of wetland quality, using wetland flora and fauna. Many indices have been developed for individual basins, or small regions within those basins, but there are few indices of wetland quality that incorporate the full range of conditions over all five lakes (i.e. Lougheed and Chow-Fraser 2002, Seilheimer and Chow-Fraser 2006) and currently there is no index of wetland quality using macrophytes that covers such a range. In chapter one of my thesis, I have developed the Wetland Macrophyte Index (WMI) to determine wetland quality for fish habitat using 127 wetlands in all five Great Lakes. I used two wetlands (Cootes Paradise marsh in Lake Ontario, Sturgeon Bay in Georgian Bay) to validate the index, where we have historical data from before and after management actions were carried out to remediate the wetlands. The WMI was also applied to two Canadian National parks on the Great Lakes (Point Pelee and Fathom Five).

Although there are numerous studies on wetland macrophytes, there is no standard method for sampling wetland macrophytes. The methods that are currently in use are designed for comparing a subset of a population to the whole population, but may not be adequate for determining the total biodiversity within a wetland. Since conservation and preservation strategies rely heavily on knowledge about endangered, threatened, rare or uncommon species, current methods of sampling for wetland macrophytes may be inadequate. In chapter two, I will compare two common plant sampling protocols (grid and transect) to a novel method called stratified sampling. I will show that the stratified method has advantages over the other two methods for determining the biodiversity of wetland macrophytes.

References

Albert, D.A., Ingram, J., Thompson, T., and Wilcox, D. 2003. Great Lakes Coastal Wetlands Classification. *Great Lakes Coastal Wetland Consortium.*

Ball, H., Jalava, J., King, T., Maynard, L., Potter, B., and Pulfer, T. 2003. The Ontario Great Lakes Coastal Wetland Atlas: A summary of information (1983-1997).

Carpenter, S.R., and Lodge, D.M. 1986. Effects of submersed macrophytes on ecosystem processes. *Aquat. Bot.*:341-370.

Chow-Fraser, P., and Albert, D.A. 1999. Biodiversity investment areas, coastal wetland ecosystems: Identification of "Eco-reaches" of Great Lakes Coastal Wetlands that have high biodiversity value. *State of the Lake Ecosystem Conference 1998.*

Cox, K.W. 1993. *Wetlands a celebration of life: A final report of the wetlands conservation task force.* Environment Canada.

Cronk, J.K., and Fennessy, M.S. 2001. *Wetland Plants: Biology and Ecology.* CRC Press, Boca Raton.

EC., and OMNR. 2003. *The Ontario Great Lakes Coastal Wetland Atlas: A summary of Information (1983-1997).*

Lougheed, V.L., and Chow-Fraser, P. 2002. Development and use of a zooplankton index of wetland quality in the Laurentian Great Lakes basin. *Ecol. App.* 12:474-486.

Mitsch, W.J., and Gosselink, J.G. 2000. *Wetlands* Third edition. John Wiley and Sons Inc, Danvers, MA.

National Wetlands Working Group. 1988. *Wetlands of Canada.* Polyscience Publications Inc, Montreal, QC.

OMNR. 1994. *Ontario wetland evaluation system: Southern manual.* Ontario Ministry of Natural Resources. Report 50254-1.

Seilheimer, T., and Chow-Fraser, P. . 2006. Development and validation of the Wetland Fish Index to assess the quality of coastal wetlands in the Laurentian Great Lakes. *Can. J. Fish. Aquat. Sci.* 63:354-366.

Smith, P.G.R., Glooschenko, V., and Hagen., D.A. 1991. Coastal wetlands of the three Canadian Great Lakes: inventory, current conservation initiatives and patterns of variation *Can. J. Fish. Aquat. Sci.* 48:1581-1594.

Wetzel, R.G. 1988. Water as an environment for plant life. *in* J. J. Symoens, editor. Vegetation of inland waters. Kluwer academic publishers,Norwell, MA, USA.

Preface

This thesis consists of two chapters that have been prepared for eventual publication in a peer reviewed journal. As the first author of these papers I collected data in the field, analyzed samples in the lab, did the statistical analysis and finally wrote the papers. Both chapters are my own original works. I am indebted to my co-author (Pat Chow-Fraser) for editing the chapters and providing her insights.

The first chapter has been submitted to the Journal of Great Lakes Research and is in revision.

Chapter 1

Use and development of the Wetland Macrophyte Index to detect water-quality impairment in fish habitat of Great Lakes coastal marshes

Melanie V. Croft and Patricia Chow-Fraser

McMaster University
Department of Biology
1280 Main Street West
Hamilton, Ontario, Canada
L8S 4K1
905.525.9140
chowfras@mcmaster.ca

Abstract

Indices have been developed with invertebrates, fish, and water-quality parameters to detect the impact of human disturbance on coastal wetlands, but a comprehensive macrophyte index of fish habitat for the Great Lakes does not currently exist. Because wetland macrophytes are directly influenced by water quality, any impairment in wetland quality should be reflected by the taxonomic composition of the aquatic plant community. We developed the Wetland Macrophyte Index (WMI) with plant presence/absence data of 127 coastal wetlands (154 wetland-years) from all five Great Lakes, using results of a Canonical Correspondence Analysis (CCA) to ordinate plant species along a water-quality gradient (CCA axis 1). We validated the WMI with data collected before and after the implementation of Remedial Actions Plans (RAPs) in Sturgeon Bay (Severn Sound) and Cootes Paradise Marsh. Consistent with predictions, WMI scores for Sturgeon Bay were significantly higher after the implementation of the RAP. Historical data from Cootes Paradise Marsh was used to track the declining condition of the plant community from the 1940s to 1990s, followed by subsequent improvements in 1997, when remedial actions had been carried out; however, when we calculated WMI scores that accounted for the presence of exotic species (WMIadj), improvements in ecological integrity of the aquatic-plant community were no longer evident. We will show how WMI scores can be used by environmental agencies to assess the historic, current and future ecological status of wetland ecosystems in two Canadian National Parks, Point Pelee National Park (PPNP) and Fathom Five National Marine Park (FFNMP).

Introduction

Coastal wetlands provide critical spawning and nursery habitat for the Great Lakes fish community (Jude and Pappas 1992, Wei *et al.* 2004) as well as valuable habitat for migratory and nesting birds (Maynard and Wilcox 1997). Approximately 60 to 80% of the coastal wetlands of the Great Lakes have been lost since the arrival of European settlers (Smith *et al.* 1991, Ball *et al.* 2003). The rapid rate of wetland loss and associated services makes it imperative that high-quality sites be identified and conserved before they are further degraded and/or destroyed. To achieve this goal, managers of environmental agencies must be provided with appropriate indicators of ecosystem health that could be used in routine monitoring programs.

Wetland degradation in the Great Lakes basin has been attributed to a variety of human disturbances, including increased loading of nutrients and sediment from agricultural and urban development (Chow-Fraser 1998, Crosbie and Chow-Fraser 1999, Lougheed *et al.* 2001), introduction of invasive species (Lougheed *et al.* 1998), and shoreline development and recreational activities (Chow-Fraser 2006). In large part, the extent to which these factors contribute to marsh degradation depends on the site type of the wetland in question. For example, coastal marshes that are located at the mouth of rivers and estuaries are particularly susceptible to altered land-uses in watersheds, and many of these in Lakes Ontario and Erie have become turbid, eutrophic systems that limit species composition of submergent macrophytes (Lougheed *et al.* 2001, McNair and Chow-Fraser 2003). In turn, changes in the submergent community are known to affect the communities of zooplankton (Lougheed and Chow-Fraser 2002), benthic invertebrates (Kashian and Burton 2000, Kostuk and Chow-Fraser 2007), and fish (Minns *et al.* 1994, Seilheimer and Chow-Fraser 2006). Because clarity of water and the amount of nutrients present in coastal marshes have such overriding influence on subsequent trophic levels, Chow-Fraser (2006) has developed the Water Quality Index (WQI) to measure the degree of degradation that can be attributed to human activities. This index includes six categories that range from highly degraded (index score of -3) to excellent (index score of +3) and has been used successfully to rank 110 wetlands throughout the Great Lakes shoreline according to their degree of water-quality impairment (Chow-Fraser 2006). WQI scores were significantly correlated with the proportion of altered (agricultural and urban) land in watersheds, and this has been confirmed as a major determinant of water-quality conditions for other Great Lakes coastal ecosystems (Danz *et al.* 2005).

Despite the effectiveness of the WQI as a monitoring tool, the effort required to measure all twelve water-quality parameters (i.e. physical characteristics, various forms of major nutrients, suspended solids and chlorophyll concentrations), makes this index unlikely to be adopted by most environmental agencies. That is one of the major reasons for the recent development of biotic indices with biota such as zooplankton (Wetland Zooplankton Index [WZI]) (Lougheed and Chow-Fraser 2002), periphytic algae (McNair and Chow-Fraser 2003), benthic invertebrates (Kostuk and Chow-Fraser 2007), and fish (Wetland Fish Index [WFI]) (Seilheimer and Chow-Fraser 2006). Interest in developing biotic indices for wetlands has been evident elsewhere as indicated by the large number of publications over the past decade (e.g. Cardinale *et al.* 1998, e.g. van Dam *et al.* 1998, Kashian and Burton 2000, Wilcox *et al.* 2002, Tangen *et al.* 2003, Uzarski *et al.* 2004).

Even though the relationship between water quality and the community of aquatic vegetation in coastal wetlands of the Great Lakes has been well studied (Lougheed *et al.* 2001, McNair and Chow-Fraser 2003, McNair 2006), no basin-wide biotic index of anthropogenic disturbance based on aquatic wetland plants for the Great Lakes has emerged. This is very surprising considering the number of advantages in using plants as a biotic indicator. Firstly, because wetland plants are essentially non-motile, their distribution can be georeferenced on each sampling occasion and changes in distribution can be tracked over time. Secondly, compared with fish surveys that require either an electrofishing boat or series of paired fyke nets (Seilheimer and Chow-Fraser 2006), plant surveys can be accomplished without specialized and expensive equipment and with only one or two trained personnel in waders and/or canoe. Unlike fish and zoobenthos surveys that require traps to be left overnight, most plant surveys can be completed within a day, during the day. Additionally, results are available immediately without the need for further processing such as required when surveying macroinvertebrates, zooplankton or periphyton.

Indices have been developed with wetland plants that have focused on upland vegetation such as the wet meadow and emergent species related to bird and mammal habitat (e.g. Wilcox *et al.* 2002, Albert and Minc 2004), but with less emphasis on the submergent, floating and aquatic emergent species that are related more to fish habitat. The development of a cost-effective index that can be used to indicate the degree of anthropogenic impact and the resultant influence on fish habitat will be an important contribution for conservation and management.

In this paper, we will show how aquatic vegetation can be used in the Wetland Macrophyte Index (WMI) to indicate human-induced degradation of coastal marshes in all five Great Lakes. The methodology for the development of the WMI is based on previous papers that relate zooplankton (Lougheed and Chow-Fraser 2002) and fish (Seilheimer and Chow-Fraser 2006) to environmental variables, using Canonical Correspondence Analysis (CCA). The use of CCA to develop plant indices is quite prevalent in Europe (eg. Dodkins *et al.* 2005) but is not yet widely used in North America. The WMI assumes that aquatic plants (all species growing obligately in flooded areas but excluding those typically associated with wet meadows) will respond directly (through competition for light and nutrients) or indirectly (through food-web interactions) to changes in water-quality conditions. We will show that this response to the degree of water-quality impairment is reflected in the taxonomic composition of the aquatic plant community. We will validate the WMI by choosing two sites that have undergone rehabilitation as part of a Great Lakes Remedial Action Plan (RAP) program (Cootes Paradise Marsh in the Hamilton Harbour RAP and Sturgeon Bay in the Severn Sound RAP (Hartig 1993)) and for which there exist plant species-lists corresponding to conditions before and after RAP initiatives. We will also show how the WMI can be used to build a monitoring program to track changes in quality of wetland habitat in two Canadian national parks that vary greatly with respect to their current ecological status. By testing the usefulness of the WMI for determining both the status of wetlands from site to site at one time, and for detecting long-term changes in a particular site over given time intervals, we will show that the WMI is versatile, easy to use, and a sensitive indicator of anthropogenic impact on coastal wetlands.

Methods

Description of Study Sites

RAP sites

The first RAP site is Cootes Paradise Marsh, which is located on the western end of Lake Ontario and is subjected to multiple stressors (Wei and Chow-Fraser 2005), including urban run-off, nutrient and sediment inputs from the Dundas Sewage Treatment Plant (see Figure 1.1) (Chow-Fraser *et al.* 1998), and feeding and spawning activity of benthivorous common carp (*Cyprinus carpio*) (Lougheed *et al.* 1998). Several initiatives were implemented to restore Cootes Paradise Marsh as part of the Hamilton Harbour RAP; a revegetation program began in 1994 (Chow-Fraser 1999b), followed by a marsh-wide carp exclusion program that began in the spring of 1997 (Lougheed *et al.* 2004, Chow-Fraser 2005). Chow-Fraser (2006) demonstrated that water quality in the marsh improved from the "Highly Degraded" category to the "Very Degraded" category from 1994 to 1998, and that by 2002, conditions were approaching "Moderately Degraded;" however, because the high nutrient and sediment concentration in the marsh is associated with both external and internal loading (Chow-Fraser 1999a, Kelton and Chow-Fraser 2005), it is unlikely that further improvements could be expected without additional human interventions.

The second RAP site is Sturgeon Bay, which is one of several bays included for remedial actions as part of the Severn Sound RAP. It is located in the southern end of Georgian Bay and was identified by the International Joint Commission as an Area of Concern (AOC) because of excessive algal growth (see Figure 1.1). A RAP was implemented in 1989 to reduce nutrient inputs into Severn Sound from eight surrounding sewage treatment plants (Sherman 2002). The Victoria Harbour Pollution Control Plant, which empties into Sturgeon Bay, began its operation as a tertiary facility in 1985. Environmental conditions have improved sufficiently that in 2003, the Severn Sound RAP was delisted.

Canadian national parks

The two Canadian national parks in this study are Fathom Five National Marine Park (FFNMP), which is located at the northern tip of the Bruce Peninsula, at the junction between Lake Huron and Georgian Bay, where development of coastal wetlands is extremely limited because of exposure to wind and wave action. The park includes several remote wetlands on Cove and Russel Islands, which receive relatively little human disturbance, compared with several on the Mainland near the town of Tobermory (see Figure 1.1). Because FFNMP has become one of the most popular destinations in Canada for freshwater recreational diving, increased tourism and cottage development in the area is threatening the integrity of the fragile coastal wetlands, especially those located on the mainland.

The other park, Point Pelee National Park (PPNP), is located on the north shore of Lake Erie, near the town of Leamington, and includes an island that has the distinction of being the southernmost point of Canada. The half a dozen wetlands of PPNP are separated by a sand barrier from Lake Erie, but are hydrologically connected to the Great Lake when high water levels cause the barrier to breach. Since the creation of the park in 1918, PPNP has been a popular destination for tourists and cottagers, as well as being a vital stop-over for migratory birds and waterfowl. The park has been impacted by agricultural run-off to the north and, from the 1940s to the 1970s, was severely affected by activities of many tourists and cottagers. Over the past three decades, managers of PPNP have gradually removed all cottages and administrative buildings, and greatly reduced vehicular use and camping within the park, to permit the wetlands and other natural areas to recover.

Description of Overall Field Monitoring Programs

Over 200 wetland-years had been sampled between 1996 and 2005 specifically for development of ecological indicators for Great Lakes coastal wetlands (e.g. Lougheed and Chow-Fraser 2002, e.g. Chow-Fraser 2006, Seilheimer and Chow-Fraser 2006). These wetlands cover a range of environmental conditions, ranging from hypereutrophic Lake Erie sites that are agriculturally impacted, to wetlands in remote islands of Georgian Bay, to several that are located in nature reserves of eastern Georgian Bay, where human impact has been minimal for over a century. Only 154 wetland-years with a complete suite of water-quality variables sampled primarily between 1998 and 2005 were chosen for development of the Wetland Macrophyte Index (Figure 1.1).

Field Sampling for Development of the WMI

All water sampling and measurements of physical and chemical parameters were conducted from a canoe or boat (depending on depth of the water). We measured temperature, pH, specific conductivity and dissolved oxygen concentration *in situ* with several types of probes. A Hydrolab™ H20 equipped with a Scout monitor (Hydrolab, Austin, Texas, USA) was used prior to 2000; during 2000 and 2001, a Hydrolab™ Minisonde and Surveyor (Hydrolab, Austin, Texas, USA) was used, and from 2002-2005 a YSI™ 6600 probe with 650 display was used (YSI, Yellow Springs, Ohio, USA). We compared all three probes in 2001 and found no significant differences for any of the above parameters (Chow-Fraser 2006). All sensors for the instruments were calibrated on a weekly basis. Daily calibration was not feasible due to the remoteness of many of our sampling sites. Sampling was always conducted during daylight hours (generally between 0900 and 2000). After 2000, coordinates of the sites were taken with either a Trimble™ GPS (4-5 m accuracy) or a Garmin™ Etrex GPS (4-6 m accuracy). For sites sampled prior to 2000, coordinates were taken from published sources (Crosbie and Chow-Fraser 1999, Lougheed and Chow-Fraser 2002).

Water was collected for nutrient and turbidity analysis in 1-L van Dorn bottles at mid-depth in water outside the submergent plant zone. Water for nutrient analysis was dispensed into clean Nalgene™ bottles (acid washed and rinsed with deionized water), while those for chlorophyll analysis was dispensed into opaque Nalgene™ bottles. All samples were stored on ice in a cooler and were analyzed later that day at the field lab. Turbidity samples were collected in an identical manner and were measured in the canoe, with a Hach™ 2100 Portalab. Methods used for processing samples in the field and the laboratory have been documented in detail elsewhere (Lougheed *et al.* 1998, Chow-Fraser 1999a; 2006). The final list of nutrient and suspended-solids variables included were: pH, total phosphorus (TP), organic suspended solids (OSS), inorganic suspended solids (ISS), total nitrate-nitrogen (TNN), total ammonia nitrogen (TAN), conductivity (COND) and chlorophyll (Chl).

Aquatic Plant Surveys

Development of the WMI

Plant data used for development of the WMI were collected between 1996 and 2005, although the majority was collected between 2000 and 2005. On each sampling occasion in a wetland, the aquatic-plant community was surveyed as follows (usually between late June and late August). In wadeable water, emergent plants would be surveyed by walking along transects parallel to the shoreline within the flooded zone. Some submergent taxa could be identified within these transects, but majority of these were surveyed with quadrats (0.75m x 0.75m) from a canoe or boat, within the vicinity where fyke nets had been set contemporaneously to survey the fish community. Depending on the size and complexity of the wetland, these surveys would take from 20 minutes to several hours to complete. Generally, 10 to 15 quadrats would be sampled in each wetland and only the occurrence of species was noted—i.e. we did not estimate percent cover of particular species within the quadrats. At least 10 quadrats were sampled in each wetland, and after that point, the sampling would cease when no new species were found in two consecutive quadrats. The focus of the survey was to identify submergent, emergent and floating plant taxa that serve as fish habitat; therefore, species associated with wet meadow were largely ignored. All plant taxa were identified with keys from Crow and Hellquist (2000) and Chaade (2002).

Comparison of submergent plant communities in Sturgeon Bay

Percent cover of submersed aquatic plants was observed at eight stations in Sturgeon Bay in late September 2004. All observations were noted by divers in the water within an area of approximately 55 m^2 (diameter of a circle with the length of a 16-ft canoe). The sites were chosen to represent different habitats within the bay based on a previous report (Sherman *et al.* 1989).

Application of the WMI to wetlands

The WMI was developed with 154 wetland-years (127 wetlands) for which we had both plant presence/absence data and water quality information. The wetlands that were used for validation (Cootes Paradise Marsh and Sturgeon Bay) and direct application (Fathom Five and Point Pelee) were excluded from the CCA to eliminate direct overlap between sites used for development and validation (details of these data will be given separately below). Following development of the WMI, we applied the index to data that were available for 176 wetlands (135 wetlands) for which we could calculate both WMI and WQI scores, since we wanted to determine the relationship between these indices. Data for this portion of our study came from wetlands that had been sampled between 1996 and 2005 according to procedures described previously. This larger database included the 154 wetland-years used in the development of the WMI, as well as 22 that had been excluded for a variety of reasons; some had been specifically excluded because they were case studies we wanted to use to validate the WMI, and others because the plants had been surveyed by inexperienced people, and we did not want to compromise the quality of the database used to develop the WMI. Application of the WMI to such a heterogeneous dataset allowed us to assess the robustness of the WMI across all environmental conditions, including differences in water-level scenarios across years, and differences due to lake-basin origin.

The historical data used to calculate the WMI values for Cootes Paradise Marsh were acquired from the Royal Botanical Gardens for 1946, 1973, 1993 and 2003 (Chow-Fraser 1998, Rothfels *et al.* 2004), while data for 1994, 1996 and 2002 were carried out by Chow-Fraser (unpub. data). No WQI values could be calculated for Cootes Paradise prior to 1993 because relevant water-quality information were lacking. Sturgeon Bay was sampled for plants in 1988 by Sherman *et al.* (1989), and 8 of those sites were re-sampled in 2004 by Sheila McNair with the same protocols as outlined above. Similar to the Cootes study, no WQI values could be calculated for the 1988 period because of the absence of appropriate water-quality data.

The WMI was calculated for various sites in PPNP and FFNMP from presence/absence data surveyed in late June, 2005, and WQI scores were determined for these sites based on water-quality information collected at the time of the plant surveys. We were not able to obtain detailed survey data from historic periods for either national park to calculate historic the WMI scores for comparison with 2005 scores.

Multivariate statistical analyses

Canonical correspondence analysis is a useful tool in ecological analysis because it produces synthetic axes that maximally separate the species niches (ter Braak and Verdonschot 1995). It uses both species and environmental data and follows the theory that each species thrives under specific environmental conditions. CANOCO™ 4.5 (ter Braak and Smilauer 1998) was used to run both the detrended correspondence analysis (DCA) and canonical correspondence analysis (CCA). Detrended correspondence analysis was initially used to verify that the species data had unimodal distributions across the environmental (water-quality) gradient. CCA was used to ordinate the species along the environmental gradient, where the ordination is constrained by the environmental variables. Environmental variables were standardized to a mean of zero and a standard deviation of one. Since CCA has the tendency to over emphasize rare species, taxa that occurred in fewer than 3 wetlands were excluded (ter Braak and Smilauer 1998). The significance of the axes was determined with the full model (499 permutations) and Monte Carlo permutations. Biplot scaling was used and the scaling was focused on interspecies distances. Points in the ordination plot were based on LC (linear combination) scores (biological data is described in relation to the environmental variables) which is the standard method in CANOCO 4.5. PC ORD™ 4.0 was used to run nonmetric multidimensional scaling (NMS) with the same dataset to verify the results of the CCA. SAS JMP IN 5.1 (SAS Institute, Cary, North Carolina, USA) was used for all other statistical analyses including: t-tests, ANCOVA and ANOVA.

Geographic information system

ArcMap 8.2 (ESRI copyright 2002) was used to produce maps, and ArcView 3.2 (ESRI copyright 1992-1999) was used to determine distances from sampling points to shore.

Results and Discussion

The 127 (154 wetland years) wetlands used for the development of the WMI were located throughout the 5 Great Lakes, on the U.S. and Canadian shorelines (Figure 1.1, Appendix 1.1). Although their ecological conditions vary widely throughout the basin, there is a general trend towards more polluted wetlands being associated with the two lower lakes, Lakes Erie and Ontario, where there is greater agricultural and urban development (USEPA and GC 1995). For comparison, we calculated mean, median, minimum and maximum values for each environmental variable on a lake-by-lake basis (Table 1.1). The most pristine wetlands in the remote areas of eastern Georgian Bay and the North Channel were associated with the lowest concentrations of nutrients and suspended solids (mean turbidity of 2.87 NTU, and mean inorganic suspended solids of 2.11 mg/L). These values are significantly lower than corresponding values for Lake Erie and Lake Ontario, where average levels of nutrients were an order of magnitude higher, and suspended solids concentrations were 7-9 times higher (see Table 1.1). Similarly, mean CHL in Georgian Bay wetlands were 10-fold lower than those in Lake Erie and Ontario (2.28 versus 24.82 and 16.37 μg/L, respectively). Mean conductivity (μS/cm) levels ranged from 126 in Georgian Bay to a high of 388 and 470 for Lakes Erie and Ontario, where many wetlands are urbanized and receive large volumes of highway runoff (e.g. Eyles *et al.* 2003). By comparison, pH values for all five Great Lakes were generally circumneutral, with median values of 7.61 for Lake Superior to slightly more alkaline conditions for Lake Michigan of 8.27. Given that most wetlands in Georgian Bay were un-impacted, it is not surprising that the corresponding mean WQI score was relatively high (1.37, signifying very good conditions) and that for Lake Erie was relatively low (-0.35, signifying moderately degraded conditions). It is also important to note that there were no wetlands in the "Excellent" category for either Lake Ontario or Lake Erie.

Development of the WMI

As a first step in the development of the WMI, we carried out a CCA with environmental and plant data from the large dataset that included 154 wetland-years (Figure 1.1). Altogether, eleven environmental variables were entered into the analysis, including latitude, longitude, turbidity, conductivity, pH, ISS, OSS, TAN, TNN, TP and CHL (refer to Table 1.1 for explanation of abbreviations). Indicators of wetland degradation such as high nutrient levels and high turbidity were found to ordinate along the first synthetic CCA axis. CCA Axis 1 explained 40.2% (P=0.002) of the variance while Axis 2 explained an additional 16.2% (P=0.002; Figure 1.2 and 1.3). The "cumulative percent variance" is a percent of the total explained variance of the species-environment relation, and should not be interpreted as the amount of variation of the community that is explained by the environmental variables. Axis 1 was highly correlated with COND (r= 0.83), TP (r=0.57), TAN (r= 0.52) and latitude (r=0.46), while Axis 2 axis was highly correlated with longitude (r=.66) and latitude (r=0.43). The cumulative eigenvalue of 3.521 indicated there was a good separation of niches for the species in our dataset.

Figure 1.2 and 1.3 are biplots of CCA Axis 1 against CCA Axis 2 for the 154 wetland years in this study. The location of a species or a site along an axis is referred to as its "centroid", and the spatial association of plant and site centroids provide very useful information. For instance, centroids located on the right side of Figure 1.2 correspond to sites that tend to have high turbidity and nutrient levels and are considered to be impacted and degraded, whereas those on the left side correspond to sites that tend to have low nutrient and turbidity levels and are considered un-impacted and healthy. In the same way, corresponding species centroids located on the left side of Figure 1.3 are less tolerant of eutrophic and turbid water, and include species such as horned bladderwort *Utricularia cornuta* (UTCO) and floating heart *Nymphoides cordata* (NMCO). By contrast, species such as fanwort *Cabomba sp* (CABO) and Greater Duckweed *Spirodela polyrhiza* (SPIR), which are found on the right side in Figure 1.3, are more tolerant of degradation. There is also a latitudinal gradient evident, because species found in the upper left quadrant were almost exclusively associated with Lake Superior (e.g. Mare's-tail *Hippuris vulgaris* [HIVU]), while species found in the lower right quadrant were more likely to be associated with Lake Ontario (e.g. Water-meal *Wolfia sp* [WOLF] and Yellow water lily *Nuphar advena* [NUAD]).

We adopted the general formula used by others (Lougheed and Chow-Fraser 2002, Seilheimer and Chow-Fraser 2006) to generate the WMI score. The two parameters, known as the optimum (U-value) and the tolerance (T-value), are related as follows:

Equation 1:

$$WMI = \left(\frac{\sum_{i=1}^{n} YiTiUi}{\sum_{i=1}^{n} YiTi} \right)$$

Where: Y_i = if the species is present, this value is 1; if absent, it is 0

T_i = value from 1-3 or niche breadth of species *i*

U_i = value from 1-5, tolerance of species *i* to degradation

We used the position of the centroid along the CCA axis 1 to determine the U-value for that species. The species centroids were ranked and sorted into five groups, and each group was assigned a value from 1 to 5, with equitable distribution of species in each group. Species with centroids that had high positive values (located to the extreme right of CCA axis 1) were given a U-value of 1 because they were associated with sites that had high nutrient and suspended solids concentrations, whereas those with high negative values (located to the extreme left of axis 1) were given a U-value of 5 because they were associated with sites with very low concentrations (Figure 1.3) (Table 1.2). All centroids located between these two extremes assumed intermediate values, depending on their location along the first axis. It is useful to think of the U-value as being an index of the species' tolerance of (or sensitivity to) degraded water-quality, such that a value of 1 indicates most tolerant and a value of 5 indicates least tolerant.

The T-value was an indication of niche breadth for each species, and was estimated from the standard deviation of the species scores from the CCA print-out. The standard deviations of the species scores were first sorted in descending order (similar to the method used to assign U values), and species with a broad niche (large standard deviation) were assigned a T value of 1, whereas species with a narrow niche (small standard deviation) were assigned a value of 3.

We confirmed the results of the CCA by running a non-metric multidimensional scaling (NMS) analysis with the species data, and found very similar results, indicating a strong underlying degradation gradient. We wanted to be sure that there was no other gradient that was governing the distribution of species (e.g. sediment composition). NMS is another ordination method, but unlike CCA, it uses only the species information to align the data according to a gradient. Similar results from a NMS analysis would confirm that there is only one strong underlying gradient. We found a highly significant correlation between NMS Axis 2 and the CCA Axis 1 scores (degradation axis) ($r^2 = 0.82$, $P < 0.0001$). When NMS axis 2 scores were sorted by magnitude and grouped into five categories in a similar fashion as had been done for the CCA Axis 1 scores (equivalent to U values), we found almost complete overlap between NMS and CCA groupings.

<u>U and T values for macrophyte taxa</u>

U and T values were determined for a total of 94 taxa, 26 emergent, 16 floating and 52 submergent (Table 2). There were 15 taxa that were identified only to genus, and U and T values assigned to these were determined in different ways. Plants such as muskgrass (*Chara*), stonewort (*Nitella*) and quillwort (*Isoetes*), that we could not readily identify to species were treated as a single taxon. On the other hand, pondweed (*Potamogeton*), milfoil (*Myriophyllum*) and bladderwort (*Utricularia*) had species with a wide range of U and T values, and we could not assign simply assign them an "average" value. Instead, we gave the most conservative values associated with the genus. Hence, the more coarse the identification, the lower the WMI score. We felt that this was less objectionable than omitting the entry, and as long as there is consistent treatment within a study, the resulting scores should be directly comparable. Therefore, the experience and knowledge of the person conducting the plant survey could affect the value of the WMI score, although we do not yet have empirical evidence of such a bias.

Three of the 26 emergent species could be considered indicative of very good conditions (U value of 5), and these included pipewort (*Eriocaulon aquaticum*), three-square bulrush (*Scirpus americanus*) and horned bladderwort (*Utricularia cornuta*), which were almost always found in the high-quality sites; except for 3-squared bulrush, they all tended to have a very narrow niche breadth. Indicators of good conditions (U value of 4) included branched burreed (*Sparganium androcladum*), softstem bulrush (*Scirpus validus*), hardstem bulrush (*Scirpus acutus*), water horsetail (*Equisetum fluviatile*), and two species of spike rush (*Eleocharis acicularis* and *E. smallii*). Species we found to be indicative of degraded water quality (U value of 1) were dominant in polluted sites and included purple loosestrife (*Lythrum salicaria*), smartweed (*Polygonum amphibium*), the two non-native cattail species (*T. angustifolia*, the putative hybrid *T.xglauca*) and the unbranched burreed (*Sparganium emersum*). Several species could be considered "neutral" in that they were cosmopolitan and seemed to be tolerant of a lot of different conditions. These included pickerelweed (*Pontederia cordata*), small arrowhead (*Sagittaria cuneata*), the giant burreed (*Sparganium eurycarpum*) and the native broadleaf cattail (*Typha latifolia*). They accounted for some of the most common species of emergent plants encountered in our dataset (Table 1.2).

Majority of the floating species were able to withstand elevated level of nutrients and turbidity. Because photosynthesis takes place above the water surface, turbidity in the water column was not a limiting factor for growth for these species. Free-floating species like the duckweeds and watermeal (*Lemna minor, Lemna trisulca* and *Wolffia sp.*) must obtain all of their nutrients from the water column since they are not rooted to the sediment; for this reason, they tended to be found in locations impacted by urban and agricultural runoff or sewage inputs. Of the 16 floating species, only two were found in high quality wetlands and these were the floating heart (*Nymphoides cordata*) (U value 5) and Water shield (*Brasenia schreberi*) (U value 4). Both of these species require high water clarity and low turbidity, and are consequently only found in wetlands with little or no human impact. Since floating heart has a very narrow niche breadth (T value of 3), it can only grow in relatively undisturbed sites, whereas the water shield has a broader niche (T value of 1) and can be found in a larger range of conditions. Both the common yellow pond lily (*Nuphar variegata*) and the fragrant white water lily (*Nymphaea odorata*) are widespread throughout the Great Lakes (56.7 and 66.5 % occurrence). Both species were tolerant of relatively degraded conditions, and were given a U value of 2, and a T value of 1, based on their ubiquitous distribution. The four exotic floating species, water hyacinth (*Eichornia crassipes*), frogbit (*Hydrocharis morsus-ranae*), water lettuce (*Pistia stratiotes*), and water chestnut (*Trapa natans*) were each assigned a U value of 1 because of their ability to invade new habitat and out compete native species.

Submersed aquatic species differ from the floating and emergent species ~~in that they~~ must spend most, and sometimes all parts of their life cycle (e.g. coontail (*Ceratophyllum demersum*) Philbrick and Les 1996) within the water column. They have different growth forms that are thought to be adaptations for living below the water surface, where light availability is often a limiting factor for growth (Middleboe and Markager 1997). Chambers and Kalff (1987) compared the growth of three species with different growth forms under different combination of sediment nutrient and light conditions. They found that biomass of the slow-growing, low-lying species, fern-leaf pondweed (*Potamogeton robbinsii*), was entirely dependent on light levels at the sediment surface, and did not require high nutrient levels since the plant does not have very large stems or leaves. By contrast, the biomass of the tall erect pondweed (white-stem pondweed, *Potamogeton praelongus*), which sends its leafy branches to the surface, was primarily determined by sediment fertility, since it requires ample nutrients to grow the large number of leaves. The biomass of the rosette species, tape grass (*Vallisneria americana*) was dependent on both sediment irradiance and the sediment fertility, since its growth form is intermediate between these two. Even though freshwater sponges are not green plants *per se*, we have included them in this major group, because their distribution is largely governed by water quality, since they are sessile, low-growing forms that require good light penetration to support algal photosynthesis. Hence, their presence in a wetland is evidence of pristine water-quality conditions (Lauer *et al.* 2001).

In this study, we could explain the U values (Table 1.2) assigned to certain species based on their growth form. For instance, the delicate rosette-forming species such as water lobelia (*Lobelia dortmanna*) and quillwort (*Isoetes sp.*) were assigned high values of 5 and 4, respectively, because they only occurred in undisturbed wetlands (only Georgian Bay and Lake Superior), where there is generally good light penetration to the sediment, and relatively infertile sediments that benefit these slow-growing rosette species (Farmer 1989, Middleboe and Markager 1997). On the other hand, species that grow quickly and form canopies near the water surface, such as the invasive Eurasian milfoil (*Myriophyllum spicatum*) and sago pondweed (*Stukenia pectinatus*) were both assigned U value of 1. These species can tolerate and thrive in wetlands with high levels of turbidity and nutrients (Chambers and Kalff 1985, Lougheed *et al.* 2001). Species such as tape grass (*Vallisneria americana*), clasping leaved pondweed (*Potamogeton richardsonii*), and slender water nymph (*Najas flexilis*), were assigned U value of 3, which appropriately reflected their intermediate growth forms (Hudon *et al.* 2000).

Reasons other than their growth form may be invoked to support why Canadian waterweed (*Elodea Canadensis*) and coontail (*Ceratophyllum demersum*) were assigned relatively low U values (2 and 1 respectively). Coontail lacks roots, and therefore, assimilates nutrients directly from the water column and can accumulate excess nitrogen early in the season (Mjelde and Faafeng 1996); hence, they do not tend to be found in undisturbed sites, but are instead dominant in polluted wetlands. Canadian waterweed, on the other hand, is a species that is apparently tolerant of shade stress, and is also a very good competitor against the Eurasian milfoil (*Myriophyllum spicatum*) (Abernethy *et al.* 1996) but does have a root system, and this seems consistent with its U value of 2. Muskgrass (*Chara* sp.) can remain green throughout the winter, is a fast colonizer, and can tolerate relatively low light levels, persisting at depths lower than those corresponding to angiosperms in clear lakes (Blindow 1992). However, in turbid systems, charophytes are light-limited, and cannot compete effectively for light against the canopy-forming species such as Eurasian milfoil. Hence, the statistically derived U-value of 3 reflects these intermediate characteristics.

Comparison of WMI scores

The WMI score for a wetland can theoretically range from a minimum of 1.00 to a maximum of 5.00. In this study, however, only four wetlands had the lowest score of 1.00, and none of the wetlands had the maximum score of 5.00. Wetlands with the minimum score included Old Woman creek, Tremblay beach, Little Cataraqui creek and Grindstone creek (see Appendix 1), which are all degraded wetlands in Lakes Erie and Ontario, and have turbid, nutrient-rich water and where all macrophyte species present were associated with the lowest U and T value of 1 (e.g. Sago pondweed). The reason that none of the wetlands had a maximum WMI score of 5.00 is because both specialist species that require good water quality (U value of 5 and T value of 3, e.g. pipewort and alternate leaf water-milfoil), as well as generalist species that can tolerate a wide range of water-quality conditions (U value of 3 and T value of 1, e.g. flat-stemmed pondweed) can be found in pristine wetlands. The maximum WMI score in this study was 4.10 (Tadenac Bay, see Appendix 1.1), which is located in a fish and wildlife sanctuary in eastern Georgian Bay, that has been managed with minimal human disturbance since the late 1900s.

Accounting for Presence of Exotic Species

In developing the WMI, our principal goal was to use the plant community to reflect water-quality conditions, primarily with respect to water clarity and nutrient concentrations because these are the usual human-induced impacts on wetland environments. However, we recognize that some changes in the species composition of the plant community are not reflective of altered water quality, but that nevertheless reflect human-induced disturbance (e.g. recreational/boating activities). McNair (2006) has shown that impacted coastal marshes tend to be more susceptible to exotic invasions than are unimpacted wetlands, and over time, native species in human-disturbed sites can lose ground to exotic species (e.g. Wei and Chow-Fraser 2006). Therefore, we wanted to account for the presence of exotic species by adding an additional term to the right of Equation 1 as follows:

Equation 2: $$WMI_{adj} = \left(\frac{\sum_{i=1}^{n} YiTiUi}{\sum_{i=1}^{n} YiTi} \right) - \sqrt[2]{Ex}$$

where "Ex" equals the proportion of floating and submergent taxa that are exotic (i.e. non-native), and we called this the adjusted the WMI (WMIadj).

Relationship Between WQI and the WMI

To aid interpretation, we have calculated a range of WMI scores that can be roughly equivalent to the six categories of water-quality conditions used by Chow-Fraser (2006; Table 3). These can be used qualitatively to indicate overall wetland conditions when no water-quality information are available. For example, wetlands with WMI scores <2.5 can be considered impaired (moderately to highly degraded conditions) and may require restoration and other management interventions. This list contains many of the wetlands in Lake Ontario and Erie that have been targeted for restoration as part of the Great Lakes Remedial Action Plans (RAP) (Cootes Paradise Marsh, Grand River, Humber River, and wetlands of Bay of Quinte), as well as wetlands that were part of the Severn Sound RAP (Matchedash Bay and Sturgeon Bay) (see Appendix 1.1 for WMI scores for all 176 wetland-years in this study). On the other hand, wetlands with scores >2.5 can be considered in "good" to "excellent" conditions, and do not show negative signs of human disturbance. This list includes wetlands from all five Great Lakes, although the majority are in eastern Georgian Bay, Huron and Superior.

To quantify the extent to which WMI scores accurately reflected water-quality conditions, we regressed the WMI scores against corresponding WQI scores for 176 wetland years from our large database that had both water-quality and plant information (Figure 1.4a). We found a highly significant linear relationship between the two indices ($r^2 = 0.57$, $P = 0.0001$), indicating very good correspondence between the presence/absence of plants and water quality conditions. As indicated in the Methods, 22 wetlands had been excluded from the WMI development (Figure 1.4b) for various reasons. The distribution of these wetlands is clearly within the range of data used for the WMI development, indicating that this index is robust, especially considering some of these sites had not been sampled as thoroughly as sites used for the development of the WMI. Data associated with the two lower lakes and their connecting channels (Erie/St. Clair, Ontario and Niagara) tended to have WQI scores ≤ 1 and WMI scores ≤ 3, and none of the wetlands in Lake Erie and Ontario were in the "excellent" category (WQI score ≥ 2). By contrast, over half of the wetlands of Georgian Bay, and many of those in Lake Huron were in the "very good" to "excellent" categories (WQI score ≥ 1), and these designations were matched by significantly higher WMI scores (3.33 vs. 1.92 and 2.12 for Georgian Bay and Lakes Ontario and Erie, respectively; ANOVA, $P<0.0001$, $n=176$).

We compared the relationship between the WMI (thick line in Figure 1.5a) and the adjusted WMI (dotted line in Figure 1.5a) with WQI scores. Lack of a significant interaction between type of WMI score and the WQI ($P=0.43$, ANCOVA) indicated that slopes for the two regression equations were statistically homogenous; we therefore compared the two intercepts and found a numerically small but statistically significant difference (2.51 vs. 2.67 for the WMI and the WMIadj , respectively). This is empirical evidence that the proportion of exotic species in the macrophyte community has a measurable effect on the WMI score, and should probably be incorporated as a metric when comparing wetlands across the Great Lakes basin.

<u>Validation of the WMI: Cootes Paradise Marsh, Hamilton Harbour RAP</u>

Long-term changes in the biotic community (plants, fish and zoobenthic invertebrates) of Cootes Paradise Marsh have been well documented (Chow-Fraser 1998, Lougheed *et al.* 2001, Lougheed *et al.* 2004, Chow-Fraser 2005). Compared with this wealth of biotic information, historic data on water-quality conditions in the marsh are scarce to non-existent (Chow-Fraser *et al.* 1998, Chow-Fraser 1999a), and this situation is not unusual for most restoration projects. This is one of the main reasons why indices based on biotic information tend to be more useful for tracking long-term changes. From previous studies, we know that the marsh ecosystem of Cootes Paradise had been severely stressed by urban and agricultural development in the watershed, sustained high water levels, and bioturbation from a very large population of common carp, which is exotic to the Great Lakes (Lougheed *et al.* 1998, Lougheed and Chow-Fraser 2001, Lougheed *et al.* 2004, Chow-Fraser 2005, Wei and Chow-Fraser 2006). One of the most visible losses through the decades has been a decline in percentage cover of emergent vegetation from over 85% in 1934 to < 15% in the last two decades (Chow-Fraser *et al.* 1998), but along with change in areal cover of cattails, there has been a concomitant decrease in species richness and diversity of submersed aquatic plants. One of the primary goals of the remedial actions was therefore to help restore the submersed plant community through a carp exclusion program that began in the spring of 1997 (Lougheed *et al.* 2004). The reason for choosing carp removal as a restoration strategy is that the spawning (Lougheed *et al.* 1998) and feeding (Chow-Fraser 1999a) activities of common carp can account for up to 35-40% of the turbidity in Cootes Paradise Marsh, and one of the expectations of the restoration was that water clarity in the marsh would improve sufficiently after carp exclusion to allow submergent vegetation to become re-established.

Prior to carp exclusion, surveys of the plant community conducted in 1994 and 1996 revealed only 4 and 5 species, respectively. Five years following remedial actions in 2002, this number had increased to 7, and a year later, it had increased to 8. However, these numbers are still low compared to the historic high of 15 during the 1940s (Chow-Fraser *et al.* 1998). Using the plant species list, we calculated the WMI scores to track changes before and after the RAP implementation. In Figure 1.6, we show a steady decline in the WMI scores through the 5 decades prior to RAP implementation that mirrored the decline in species richness already noted. WMI scores corresponding to the two data points following carp exclusion (2002 and 2003) were much higher, and this confirms the measurable improvement in water quality of Cootes Paradise noted by Chow-Fraser (2006) based on WQI scores. Because both water-quality and plant information were available for 1994, 1996 and 2002, we calculated pairs of the WMI and WQI scores, and superimposed these (open squares in Figure 1.5b) on the regression line for the 176 wetlands. This comparison indicates that the WMI can be applied successfully to track the restoration of Cootes Paradise Marsh since all of the Cootes data fell within the 95% confidence intervals of the regression line.

Even though the 2002 and 2003 WMI scores were higher than that in 1996, the increase was due to the presence of several exotic species, including the Eurasian milfoil (*Myriophyllum spicatum*), and the ornamental water lettuce (*Pistia stratoites*) both of which are invasive and thrive in warm, fertile waters (Cofrancesco 1998, Gordon 1998). Another non-native species, *Potamogeton crispus*, was also found in 2003, but it had already been observed in the 1946 and 1972 surveys. We accounted for the presence of these exotics by calculating the WMIadj score for each year and found that the values had decreased to pre-RAP levels calculated for the early1990s (see Figure 1.6, open circles). Therefore, although the trend in the WMI indicated an overall improvement in water quality, the trend in the WMIadj revealed that the ecosystem health of the wetland continues to be poor.

Validation of the WMI: Sturgeon Bay, Severn Sound RAP

Unlike Cootes Paradise Marsh, very little published data exist that can be used to track changes in the environmental quality of Sturgeon Bay before and after the RAP. Sherman (2002) reported historical total phosphorus concentrations for Sturgeon Bay, which were obtained from Environment Canada. The relationship between increased nutrients (especially phosphorus) leading to increased phytoplankton growth, resulting in increased turbidity and decreased submergent aquatic vegetation has been well established in the literature (Hough *et al.* 1989, Crowder 1991, Golterman 1995, Lougheed *et al.* 2001, McNair and Chow-Fraser 2003). Nicholls (1988) reported high total phosphorus levels in the vicinity of Sturgeon Bay in Severn Sound between 1973-1982, which resulted in high amounts of phytoplankton that negatively impacted the submersed aquatic vegetation. When data were grouped before and after 1985 (when the Sewage Treatment Plant had been built in Victoria Harbour), we found that the post-1985 mean (for data from 1986 to 2003) was significantly lower than that for the pre-1985 period (data from 1970 to 1984) (19.50 μg/L and 16.26 μg/L respectively; t-test, P=0.0317) (see Figure 1.7), indicating that overall water-quality conditions have indeed improved.

Based on the reduction in TP concentrations (Figure 1.7), we expected to find corresponding improvements in WMI scores. The most comprehensive plant survey conducted in Sturgeon Bay prior to implementation of the Severn Sound RAP was carried out in 1988 by Sherman (1989). He collected plant species information at 15 sites in Sturgeon Bay in 1988; eight of these sites were revisited by us during 2004, and similar information was collected (Table 1.4). Two striking changes took place over the 16 years. Eurasian milfoil (*Myriophyllum spicatum)* had almost disappeared from Sturgeon Bay by 2004, even though it had been a dominant component in at least 3 stations during 1988, and was known to be a dominant species in two earlier surveys of the entire Bay (1980 and 1982, cited in Sherman *et al.* 1989). It appeared to have been displaced by a conspecific (and likely competitor), the common milfoil (*M. sibiricum*), which was found abundantly in at least half of the sites during 2004, despite its rare occurrence during the 1988 survey. Several species were also found in greater abundance in 2004, including flat-stemmed pondweed (*P. zosteriformis*), large-leaved pondweed (*P. amplifolius*), white-stemmed pondweed (*P. praelongus*), and water marigold (*B. beckii*). Also noteworthy is the complete disappearance of the exotic species, curly-leaf pondweed (*P. crispus*), coupled with the establishment of the freshwater sponge at two sites.

We compared the species richness of plants between the two surveys, and found that on average, there were 10 ± 1.51 species observed per site in 1988, compared with only 7.87 ± 2.1 in 2004 and this difference was statistically significant (Paired T-test; P=0.0358) (Table 1.4). However, comparison of the Shannon-Weiner Index Scores revealed no significant difference between years (Paired t-test; P=0.83) (Table 1.4). We then generated the WMIadj scores for each site, using the 1988 and 2004 data (Figure 1.8), and found a significant improvement in the WMI scores over the 16 years (Paired t-test; P < 0.0012). Although the extent of improvement seemed to depend on the total distance separating a particular site from the sewage outflow pipe, we found no statistical evidence to support this conclusion; however, there was a general trend towards larger improvements for sites located closest to the sewage outflow (Table 1.4). We also investigated if the extent of improvement from 1988 to 2004 could be related to distance from shore (calculated with a GIS), but this also proved inconclusive (t-test; P = 0.384). Hence, variability associated with extent of improvement from site to site could not be attributed to either distance from sewage outflow pipe or to shoreline impacts.

Use of the WMI in Routine Monitoring Programs

Two national parks were used as case studies to demonstrate the usefulness of the WMI in routine monitoring programs. Within each of the parks were a series of wetlands, some more impacted than others with respect to past, current and potential human-induced disturbance (see Site Description above). There were 4 wetlands in Fathom Five National Marine Park (FFNMP): Boat Passage (BG), Hay Bay 1 (HB1), Hay Bay 2 (HB2), and Cove North (CN) (see Figure 1.5b). Likewise, there were 4 wetlands in Point Pelee National Park (PPNP): West Cranberry (WC), East Cranberry (EC), Sanctuary Pond (SN) and Big Pond 1 (BP1) (Table 1.6).

All of the wetlands in these two parks had been surveyed for plants from a canoe on only one occasion during the summer of 2005 (see Table 1.6). To approximate the effort most likely afforded by environmental agencies, wetland surveys were carried out by two people and did not exceed half a day. Since Cove North and Boat Passage are both located on Cove Island, which is relatively free of human impact other than recreational boating, these sites should be associated with very high WMI scores. By comparison, Hay Bay 1 and 2 are located on the mainland and are vulnerable to sediment and nutrient enrichment resulting from cottage development and recreational activities. We therefore expected the WMI scores to be highest in Cove North, and lowest in Hay Bay 1, which is known to support the highest level of human use.

Unlike the coastal wetlands of FFNMP, those of PPNP are not hydrologically connected to a Great Lake, because a barrier-beach on the east side of the park prevents complete mixing of the pond water with Lake Erie water. Known breaching events occurred in 1972, 1975, 1983, 1986 and 1989 (Chow-Fraser, unpub. data). Hence, during years when the barrier is breached, the marsh elevation approximates that of Lake Erie, and during these breaching events, less nutrient-rich water of Lake Erie tends to dilute the pond water, while benthivorous fish such as common carp and bullheads can invade from the lake (Beak 1988). The unique hydrology of these ponds should result in better water quality in East Cranberry Pond and Big Pond (which are more vulnerable to these breaching events), while Sanctuary Pond is expected to have the most degraded conditions because it has been hydrologically disconnected from the rest of the ponds as well as from Lake Erie for at least several decades (Chow-Fraser, unpub. data).

We found a general increase in the WMI scores with improvement in corresponding WQI scores, and when these were superimposed on the regression line for the 176 wetland-years, all of the data points were bracketed by the 95% confidence intervals of the regression line (Figure 1.5b). As expected, the lowest WMI score was associated with Sanctuary Pond, but we were surprised to find that the highest score was associated with Boat Passage rather than Cove North. We attribute the lower than expected WMI in Cove North to disturbance resulting from its geomorphology and its exposure to high wind and wave action, which are factors that negatively affect the establishment of submersed aquatic vegetation (Wei and Chow-Fraser 2007).

General Discussion

Two multivariate analyses (CCA and NMS) were used in this study to derive an index that utilizes presence/absence of wetland plant information to indicate the water-quality conditions of 154 coastal wetlands (Figure 1.1 and 1.2). The convergence of results from both analyses makes us confident in assigning the U and T values for the 94 taxa in Table 1.2. Using data from 176 wetlands throughout the five Great Lakes, we have established a highly significant relationship between the WMI and WQI scores (Figure 1.4), and this bears out our assumption that plants are indeed very good indicators of water-quality conditions in wetlands. Besides those publications cited from our own research (Lougheed *et al.* 2001, McNair and Chow-Fraser 2003), the dependence of submergent plant colonization on nutrient and turbidity levels has been documented by many others (Chambers and Kalff 1987, Hough *et al.* 1989, Barko *et al.* 1991, Golterman 1995, Tracy *et al.* 2003), and therefore we expect the results of this study to be widely applicable to wetlands in other jurisdictions. Within the Great Lakes basin, we are satisfied that the WMI can be used to rank the degree of human-induced disturbance among a wide range of coastal wetlands. However, we do acknowledge the disproportionate representation of wetlands within the Canadian portion of the shoreline (due to logistical constraints), and we recommend that a similar project be mounted to sample U.S. wetlands to address this imbalance.

The WMI was validated with historical data from Cootes Paradise Marsh and Sturgeon Bay. For Cootes Paradise Marsh, there had been sufficient water-quality information to directly compare conditions before and after RAP implementations. Chow-Fraser (2005) found a significant improvement in all water-clarity variables (extinction coefficient, Secchi depth transparency, water turbidity, and the concentration of total inorganic suspended solids) following carp exclusion at the two open-water sites (LT1 and LT5; Table 1.1 in Chow-Fraser, 2005). For instance, at LT1, water turbidity dropped 40%, from 72.2 (mean of data for 1993-1996 inclusive) to 43.6 NTU (mean of data for 1997-2001 inclusive); a similar magnitude in reduction was noted for LT5. Chow-Fraser (2006) reported a corresponding increase in WQI scores of >30%, from -2.204 in 1994, to -2.094 in 1998, to -1.498 in 2000. We were therefore pleased to see that the WMI scores were able to reflect this improvement in water clarity (a low of 1.755 in 1996 to 2.19 in 2002; Figure 1.5b), although this only represented an increase of 25%. More importantly, we want to point out that the increased WMI values after 2000 was partly attributed to the presence of two invasive exotic species (Eurasian milfoil and water lettuce), and therefore, despite the improved water quality in Cootes Paradise, the overall health of the marsh is still compromised (corresponding the WMIadj were <1.80; Figure 1.6).

Lundholm and Simser (1999) indicated that the lack of a seed bank is not a contributing factor to the return of submergent species because the majority of species historically found in Cootes Paradise Marsh are perennials that reproduce vegetatively. But it is unknown how long rhizomes and turions can persist in the sediment while they wait for favourable conditions. Prolonged periods of unfavourable conditions may prevent species from rebounding when conditions improve. Only one species found in Cootes Paradise in 1997 was annual (horned pondweed (*Zanichellia palustris*), but this pioneer species did not become abundant, probably because other more invasive and aggressive species such as *Myriophyllum spicatum*, *Pistia stratoites* and *Potamogeton crispus* benefited disproportionately from improved conditions. *Hysteresis* is the inability of an ecosystem to rebound to its previous state once the external forcing function (e.g. phosphorus enrichment) has been removed, and this has been well documented for shallow lakes in Europe (e.g. Janse *et al.* 1998, Van Nes *et al.* 2002, Zhang *et al.* 2003). This is likely the reason for the retarded improvement in the plant community of Cootes Paradise following decrease in water turbidity resulting from carp exclusion. Zhang *et al.* (2003) suggested that a shift back to clear water, macrophyte-dominated systems will be prevented in hypereutrophic shallow systems because of high release of phosphorus accumulated in the sediment, and this is consistent with findings of Kelton and Chow-Fraser (2005) for Cootes Paradise Marsh.

One of the greatest advantages of the WMI over the WQI is that there is generally historic plant information to calculate the former, but insufficient water-quality information in historic databases to calculate the latter. This was certainly true for Sturgeon Bay, for which there were many gaps in the historic water-quality database. It was difficult to find anything other than TP concentrations (Figure 1.7), and given the high interannual variation, we had to include data from 30 years to demonstrate a significant change before and after the operation of the Sewage Treatment Plant in Victoria Harbour. By comparison, calculation of the WMI only required data from two years, 1988 prior to the initiation of the Severn Sound RAP, and 2004, a year following the delisting of the RAP. Even though the 1988 plant survey had not been conducted with the WMI in mind, the data were used to generate a set of the WMI scores with relative ease (Table 3). We were able to replicate the 1988 sampling protocol during the 2004 survey to generate a corresponding set of the WMI scores (Table 1.5; Figure 1.8). Mean scores associated with the 2004 survey were significantly higher than those associated with the 1988 survey (2.96 vs. 2.42, respectively), thus independently confirming the results of the TP comparisons (Figure 1.7). Had macrophyte data been available from the early 1980s, we would probably have seen a greater improvement in the WMI values, since the 1988 survey took place three years following the start of the wastewater treatment facility in Victoria Harbour.

Another advantage to using macrophytes is that the plant community integrates effects of many factors that act concurrently on the assemblage over a long period of time (Wei and Chow-Fraser 2006). Thus, routine monitoring programs such as those required by the PPNP and FFNMP could use the WMI as a relatively cost-effective way to screen ecosystems for evidence of human disturbance, and then follow up with a more targeted and intensive sampling for water-quality conditions. We have demonstrated that the relationship between the WMI and WQI is very robust (Figure 1.5b), and that over a wide range of conditions, the WMI has been useful for ranking wetlands according to degree of human-induced degradation.

Several indices have been developed based on plant communities in coastal wetlands of the Great Lakes (Wilcox *et al.* 2002, Albert and Minc 2004, Minc and Albert 2004), but these have focused more heavily on emergent and wet-meadow taxa. Wilcox (2002) developed an index of biotic integrity (IBI) using information on fish, invertebrates and plants, and these were used to rank wetlands in Lakes Superior, Huron and Michigan, according to a gradient of human disturbance. They did not find a significant relationship between IBI scores and human disturbance, probably because of their small sample size (only 6 wetlands in each Great Lake). Simon *et al.* (2001) also developed an IBI for 18 palustrine and riverine wetlands in Lake Michigan, but it has yet to be validated and tested (i.e. the IBI had not yet been independently applied to other wetlands in the region to see whether it was useful). Compared with Simon et al's IBI, the WMI is much more suitable for assessing fish habitat, since the 100-m transect employed by Simon extended from the floodplain (wet meadow) to the littoral zone, and included areas that could not have been accessed by fish, and this likely explains the relatively small number of submergent and floating taxa (14 and 4, respectively) compared with ours (50 and 16, respectively; Table 1.2).

Wilcox *et al.* (2002) found that their IBI was influenced by low water levels experienced during the single year they collected the data, and suggested that such an IBI should only be applied to data collected under similar water-level conditions. Since the WMI was developed with data collected over 9 years (1996-2005) from all five Great Lakes, any biases due to water-level effects would have been accounted for. We are allowing the attributes of individual species and groups of species to indicate wetland quality, rather than relying on total species diversity, or total number of native taxa encountered during a particular visit. Accordingly, we are seeing a better correlation between the WMI scores and degree of human disturbance as measured by WQI scores.

Wilcox *et al.* (2002) also suggested that separate indices be made for each lake and each geomorphic type. This was not possible in our development of the WMI because the type of multivariate analyses we use requires a minimum of 40 to 50 wetlands for each lake-type, and our database (as large as it is) is too small for such rigorous statistical treatment. We also feel that such a lake-by-lake approach puts too great an emphasis on the influence of micro-climatic and geomorphic factors, and would lead to truncated gradients of human disturbance in the less populated regions of Lakes Huron and Superior. We have demonstrated that the response of the common plant species to levels of suspended solids and nutrients is similar across all five Great Lakes. The higher WMI scores associated with eastern Georgian Bay, and the lower scores with Lakes Erie and Ontario, are primarily reflective of the degree of human impact, and not to regional differences in geology or climate. When wetlands of Georgian Bay are subjected to disturbance from agricultural and recreational activities (e.g. Lily Pond in Honey Harbour, or Matchedash Bay of Severn Sound), they acquired plant species that were indicative of human-induced disturbance encountered in wetlands of the two lower lakes (see WMI scores in Appendix 1.1).

Another major difference between the WMI and previous indices (e.g. IBI of Wilcox *et al.* (2002), Minc and Albert (2004)) is that our index focuses on species related to fish habitat (submergent, floating and emergent) or taxa found in the open-water areas of wetlands, whereas the others focused on the emergent and wet-meadow communities (only 22 of the 157 transects in Minc and Albert were conducted in the submergent vegetation zone). We feel that the WMI is a more appropriate indicator of fish habitat in wetlands, whereas the other indices may be better indicators of bird habitat. If this is true, then a holistic view of wetland health would require the use of both types of indices, or development of an integrated index that includes equal treatment of all the vegetation zones.

As with other indices that rely on accurate identification of plants to the species level, the expertise of the person conducting the plant survey may have great influence on the final WMI score. We are now conducting a study to empirically determine the extent to which level of expertise of the technician will affect wetland scores, and this should provide guidance for agencies that require the use of volunteers in their monitoring programs. The volunteer Marsh Monitoring Program was started in 1994 and is currently being used by Environment Canada to determine wetland quality using bird and amphibian populations in wetlands. The marsh birds and amphibians are generally identified by sound, so volunteers must be familiar with many bird and amphibian calls and have the skills to identify them in the field. This poses a problem because generally only very experienced individuals are able to provide reliable data. The advantage of using plants for a volunteer monitoring program is that it is relatively easy to learn to identify the plant species included in the WMI, and voucher samples can be preserved for a brief period of time until they can be correctly identified by an expert.

We found both the WMI and the WMIadj useful for monitoring wetlands, and this is in agreement with the IBI of Wilcox *et al.* (2002). We recommend the use of the WMI to track effects of pollutants in wetlands since it is a sensitive indicator of water-quality conditions; however, the adjusted WMI should be used if there is an additional objective of determining the ecological health of the wetland, and to account for the impact of invasive exotic species. For example, in parks such as FFNMP, where current water-quality conditions is still excellent, increased human disturbance through increased boat traffic is more likely to threaten the biodiversity of native species rather than water quality. Parks Canada would benefit from tracking the negative impact of exotic invaders on ecosystem integrity through monitoring changes in WMIadj scores. As always, we leave it to the managers to decide which of the WMI or WMIadj index is more appropriate for their location and application.

The WMI has been developed specifically for coastal systems that have a hydrological linkage to a very large lake or bay. The lower than expected WMI score associated with Cove North in FFNMP (Figure 1.5b) indicates that this index may be modified to account for exposure disturbance due to wave and wind action. We may also be able to identify species that are more indicative of exposure or that are tolerant of different water-level scenarios. Application of the WMI to systems that are no longer hydrologically connected to the Great Lakes may also lead to lower than expected scores, and this should be the focus of a future study. We encourage others to apply the WMI to other systems (e.g. inland lakes), so that it can be further validated and/or modified to suit other purposes.

One of the main objectives of this study is to develop an index that can be used by environmental agencies that have very limited resources and personnel, and who must choose indicators that are easy to implement, cost-effective and sensitive to annual changes in wetland conditions. We believe that the WMI is such an index, and we recommend it to monitoring agencies such as Parks Canada who need to track the impact of human-induced disturbances and its effect on fish habitat in coastal wetlands in FFNMP and PPNP. Once plant sampling for the WMI is completed (usually within a few hours for most wetlands), U and T values (Table 1.2) can be applied to the data to calculate a WMI score, which can then be related back to the degree of water-quality impairment of fish habitat (Table 1.3). There is an additional bonus in that historic species lists can be used to generate WMI scores to track long-term changes in wetlands. There are a limited number of high-quality wetlands along the Great Lakes shoreline, and many of these exist in eastern and northern Georgian Bay (Chow-Fraser 2006). These wetlands provide vital spawning and nursery habitat for fish, and immediate action must be taken to ensure that they are properly assessed and inventoried before they are claimed for recreational and urban development. We hope that the WMI will emerge as one of the useful biotic assessment tools available for use by both volunteers and government personnel to monitor the health of these unique wetlands.

Acknowledgments

The development of the WMI is a labour of love, one based on data collected by a great team of students and researchers over nearly a decade. Majority of the sites were visited between 1996 and 2005 by the following in chronological order: Barb Crosbie, Vanessa Lougheed, Sheila McNair, Dennis Albert, and Melanie Croft. We thank Keith Sherman for providing data to test the applicability of the WMI for Sturgeon Bay. Funding for these sampling programs were provided by the NSERC Discovery Grant, a Fisheries Research Grant from the Great Lakes Fishery Commission, a research contract from Parks Canada, and a research grant from the Canada-Ontario Agreement through the Ontario Ministry of Natural Resources. We are also indebted to the Georgian Bay Association (GBA) Foundation for permitting us to use the Baykeeper to sample wetlands in Georgian Bay from 2003 to 2005, and for Mary Muter from the GBA who provided logistical assistance and recruited volunteer hosts to provide accommodation for the field crew while sampling in Georgian Bay. Much of the data and background information for PPNP and FFNMP were obtained from the senior undergraduate theses of Roxanne Razavi and Maja Cvetkovic, who completed their projects at the McMaster Biology Department under the supervision of PC-F.

References

Abernethy, V. J., Sabbatini, M. R., and Murphy, K. J. 1996. Response of Elodea canadensis Michx. and Myriophyllum spicatum L. to shade, cutting and competition in experimental culture *Hydrobiologia* 340:219-224.

Albert, D. A., and Minc, L. D. 2004. Plants as regional indicators of Great Lakes coastal wetland health *Aquat. Ecosystem Health and Management.* 7:233-247.

Ball, H., Jalava, J., King, T., Maynard, L., Potter, B., and Pulfer, T. 2003. The Ontario Great Lakes Coastal Wetland Atlas: A summary of information (1983-1997).

Barko, J. W., Gunnison, D., and Carpenter, S. R. 1991. Sediment interactions with submersed macrophyte growth and community dynamics. *Aquat. Bot.* 41:41-65.

Beak Consultants 1988. Marsh fish management Plan Point Pelee National Park. Implementation requirements for creel census, angler surveys and ecological monitoring Beak Consultants for Parks Canada. Report MFMP-88.

Blindow, I. 1992. Decline of charophytes during eutrophication: comparison with angiosperms *Freshwater Biology* 28:9-14.

Cardinale, B. J., Brady, V. J., and Burton, T. M. 1998. Changes in the abundance and diversity of coastal wetland fauna from the open water/macrophyte edge towards shore. *Wetlands Ecol. Manage.* 6:59-68.

Chaade, S. W. 2002. *A Great Lakes Wetland Flora: A complete guide to the aquatic and wetland plants of the upper Midwest.* Second edition. Pocket Flora Press., Laurium, Michigan. .

Chambers, P., and Kalff, J. 1985. The influence of sediment composition and irradiance on the growth and morphology of Myriophyllum spicatum *Aquat. Bot.* 22:253-263.

Chambers, P. A., and Kalff, J. 1987. Light and nutrients in the control of aquatic plant community structure. I. In Situ experiments. *J. Ecol.* 75:611-619.

Chow-Fraser, P., Lougheed, V. L., Crosbie, B., LeThiec, V., Simser, L., and Lord, J. 1998. Long-term response of the biotic community to fluctuating water levels and changes in water quality in Cootes Paradise Marsh, a degraded coastal wetland of L. Ontario. . *Wetland Ecol. Manage.* 6:19-42.

Chow-Fraser, P. 1999a. Seasonal, interannual, and spatial variability in the concentrations of total suspended solids in a degraded coastal wetland of Lake Ontario. *J. Great Lakes Res.* 25:799-813.

_____. 1999b. Volunteer-based experimental planting program to restore Cootes Paradise Marsh, an urban coastal wetland of L. Ontario *Lakeline* 19.

_____. 2005. Ecosystem response to changes in water level of Lake Ontario marshes: lessons from the restoration of Cootes Paradise Marsh *Hydrobiologia* 539:189-204.

_____. 2006. Development of the Wetland Water Quality Index for assessing the quality of Great Lakes coastal wetlands. Pages 137-166 *in* Simon, T., andStewart, T. P., editors. *Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat and Indicators*. Author House,Bloomington, IN, USA.

Chow-Fraser, P., Lougheed, L., Le Thiec, V., Crosbie, B., Simser, L., Lord, J. 1998. Long-term response of the biotic community to fluctuating water levels and changes in water quality in Cootes Paradise Marsh, a degraded wetland of Lake Ontario. *Wetland Ecol. Manage.* 6:19-42.
Cofrancesco, A. F., Jr. 1998. Overview and future direction of biological control technology. *J. Aquat. Plant Manage.* 36:49-53.
Crosbie, B., and Chow-Fraser, P. 1999. Percent land use in the watershed determines the water- and sediment-quality of 21 wetlands in the Great Lakes basin. *Can. J. Fish. Aquat. Sci.* 56:1781-1791.
Crow, G. E., and Helquist, C. B. 2000. *Aquatic and Wetland Plants of Northeaster North America: Volume one and two* The University of Wisconsin Press, Madison, Wisconsin.
Crowder, A., Painter, D.S. 1991. Submerged macrophytes in Lake Ontario: current knowledge, importance, threats to stability and needed studies. *Can J. Fish. Aquat Sci.* 48:1539-1545.
Danz, N. P., Regal, R. R., Niemi, G. J., Brady, V. J., Hollenhorst, T., Johnson, L. B., Host, G. E., Hanowski, J. M., Johnston, C. A., Brown, T., Kingston, J., and Kelly, J. R. 2005. Environmentally stratified sampling design for the development of Great Lakes environmental indicators. *Env. Monitor. Assess.* 102:41-65.
Dodkins, I., Rippey, B., and Hale, P. 2005. An application of canonical correspondence analysis for developing ecological quality assessment metrics for river macrophytes. *Freshwater Biology* 50:891-904.
Eyles, N., Doughty, M., Boyce, J. I., Meriano, M., and Chow-Fraser, P. 2003. Geophysical and sedimentological assessment of urban impacts in a Lake Ontario watershed and lagoon: Frenchman's Bay Pickering, Ontario. *Geosci. Can.* 30:115-128.
Farmer, A. M. 1989. Biological flora of the British Isles. Lobelia dortmanna. *J. Ecol.* 77:1161-1173.
Golterman, H. L. 1995. The labyrinth of nutrient cycles and buffers in wetlands: results based on research in the Camargue (Southern France) *Hydrobiologia* 315:39-58.
Gordon, D. R. 1998. Effects of invasive, non-indigenous plant species on ecosystem processes: lessons from Florida. *Ecol. App.* 8:975-989.
Hartig, J. H. 1993. Toward integrating remedial-action and fishery-management planning in Great Lakes Areas of Concern *Great Lakes Fish. Comm.*:34 pp.
Hough, R. A., Fornwall, M. D., Negele, B. J., Thompson, R. L., and Putt, D. A. 1989. Plant community dynamics in a chain of lakes: principle factors in the decline of rooted macrophytes with eutrophication *Hydrobiologia.* 173:199-217.
Hudon, C., Lalonde, S., and Gagnon, P. 2000. Ranking the effects of site exposure, plant growth form, water depth and transparency on aquatic plant biomass *Can. J. Fish. Aquat. Sci.* 57:31-42.
Janse, J. H., Van Donk, E., and Aldenberg, T. 1998. A model study on the stability of the macrophyte-dominated state as affected by biological factors. *Water Res.* 32:2696-2706.

Jude, D. J., and Pappas, J. 1992. Fish utilization of Great Lakes coastal wetlands. *J. Great Lakes Res.* 18:651-672.

Kashian, D. R., and Burton, T. M. 2000. Comparison of macroinvertebrates of two Great Lakes coastal wetlands: testing potential metrics for an Index of Ecological Integrity. *J. Great Lakes Res.* 26:460-481.

Kelton, N., and Chow-Fraser, P. 2005. Predictive factors of phosphorus release rates from hypereutrophic marsh sediments: estimating internal loading in Cootes Paradise Marsh *Lakes and Reservoir Management* 21:223-230.

Kostuk, K., and Chow-Fraser, P. 2007. Use of zoobenthos to indicate human-induced disturbance and degree of exposure disturbance in Great Lakes Coastal Wetlands *J. Great Lakes Res.* This volume.

Lauer, T. E., Spacie, A., and Barnes, D. K. 2001. Distribution and habitat preferences of freshwater sponges (Porifera) in four Southern Lake Michigan harbors. *Amer. Midland Naturalist* 146:243-253.

Lougheed, V. L., Crosbie, B., and Chow-Frâser, P. 1998. Predictions on the effect of common carp (Cyprinus carpio) exclusion on water quality, zooplankton and submergent macrophytes in a Great Lakes wetland. *Can. J. Fish. Aquat. Sci.* 55:1189-1197.

Lougheed, V. L., and Chow-Fraser, P. 2001. Spatial variability in the response of lower trophic levels after carp exclusion from a freshwater marsh. *J. Aquat. Ecosyst. Stress Recov.* 9:21-34.

Lougheed, V. L., Crosbie, B., and Chow-Fraser, P. 2001. Primary determinants of macrophyte community structure in 62 marshes across the Great Lakes basin. *Can. J. Fish. Aquat. Sci.* 58:1603-1612.

Lougheed, V. L., and Chow-Fraser, P. 2002. Development and use of a zooplankton index of wetland quality in the Laurentian Great Lakes basin. *Ecol. App.* 12:474-486.

Lougheed, V. L., Theÿsmeÿer, T., Smith, T., and Chow-Fraser, P. 2004. Carp exclusion, food-web interactions, and the restoration of Cootes Paradise Marsh. *J. Great Lakes Res.* 30:44-57.

Lundholm, J. T., and Simser, W. L. 1999. Regeneration of submerged macrophyte populations ina disturbed Lake Ontario coastal marsh. *J. Great Lakes Res.* 25:395-400.

Maynard, L., and Wilcox, D. 1997. Background Report on Great Lakes wetlands. US EPA and Environment Canada.

McNair, S. A., and Chow-Fraser, P. 2003. Change in biomass of benthic and planktonic algae along a disturbance gradient for 24 Great Lakes coastal wetlands. *Can. J. Fish. Aquat. Sci.* 60:676-689.

McNair, S. A. 2006. The use of primary producers for assessing and monitoring aquatic habitat quality in Great Lakes coastal wetlands. McMaster University, Hamilton, ON.

Middleboe, A. L., and Markager, S. 1997. Depth limits and minimum light requirements of freshwater macrophytes *Freshwater Biology* 37:553-568.

Minc, L. D., and Albert, D. A. 2004. Multi-Metric Plant-Based IBIs for Great Lakes Coastal Wetlands Michigan Natural Features inventory.
Minns, C. K., Cairns, V. W., Randall, R. G., and Moore, J. E. 1994. An Index of Biotic Integrity (IBI) for fish assemblages in the littoral zone of the Great Lakes' Areas of Concern. *Can. J. Fish. Aquat. Sci.* 51:1804-1822.
Mjelde, M., and Faafeng, B. A. 1996. Ceratophyllum demersum hampers phytoplankton development in some small Norwegian lakes over a wide range of phosphorus concentrations and geographical latitude. *Freshwater Biology* 37:355-365.
Nicholls, K. H., Robinson, G. W., Taylor, R., and Carney, E. C. 1988. Phytoplankton and phosphorus in southern Georgian Bay, 1973-1982, and implications for phosphorus loading controls. *Hydrobiologia* 163:85-93.
Philbrick, C. T., and Les, D. H. 1996. Evolution of aquatic angiosperm reproductive systems *Bioscience* 46:813-826.
Rothfels, C. J., Spisani, S. R., and Sylvester, J. L. 2004. Cootes Paradise Submergent Plant monitoring report: 2003. Royal Botanical Gardens.
Seilheimer, T., and Chow-Fraser, P. 2006. Development and validation of the Wetland Fish Index to assess the quality of coastal wetlands in the Laurentian Great Lakes. *Can. J. Fish. Aquat. Sci.* 63:354-366.
Sherman, R. K., Speller, S. D., and Pope, R. J. 1989. 1988 Survey of Aquatic Macrophytes and Benthos in Sturgeon Bay and Penetang Bay. Environment Canada and Ontario Ministry of the Environment.
Sherman, R. K. 2002. Severn Sound Remedial Action Plan Stage 3 Report: The status of restoration and delisting of Severn Sound as an Area of Concern. Environment Canada and Ontario Ministry of the Environment.
Simon, T. P., Stewart, P. M., and Rothrock, P. E. 2001. Development of multimetric indices of biotic integrity for riverine and palustrine wetland plant communities along Southern Lake Michigan. *Aquat. Ecosystem Health and Manage.* 4:293-309.
Smith, P. G. R., Glooschenko, V., and Hagen., D. A. 1991. Coastal wetlands of the three Canadian Great Lakes: inventory, current consërvation initiatives and patterns of variation *Can. J. Fish. Aquat. Sci.* 48:1581-1594.
Tangen, B. A., Butler, M. G., and Ell, M. J. 2003. Weak correspondence between macroinvertebrate assemblages and land use in prairie pothole region wetlands, U.S.A. *Wetlands* 23:104-115.
ter Braak, C. J. F., and Verdonschot, F. M. 1995. Canonical correspondence analysis and related multivariate methods in aquatic ecology. *Aquat. Sci.* 57:153-186.
ter Braak, C. J. F., and Smilauer, P. 1998. *CANOCO Reference Manual and User's Guide to Canoco for Windows: Software for canonical community ordination (version 4).* . Microcomupter Power, Ithaca, New York.
Tracy, M., Montante, J. M., Allenson, T. E., and Hough, R. A. 2003. Long-term responses of aquatic macrophyte diversity and community structure to variation in nitrogen loading *Aquat. Bot.* 77:43-52.
USEPA, and GC. 1995. *The Great Lakes: an environmental atlas and resource book.* Third edition.

Uzarski, D. G., Burton, T. M., and Genet, J. A. 2004. Validation and performance of an invertebrate index of biotic integrity for Lakes Huron and Michigan fringing wetlands during a period of lake level decline. *Aquat. Ecosystem Health and Management* 7:269-288.
van Dam, R. A., Camilleri, C., and Finlayson, C. M. 1998. The Potential of Rapid Assessment Techniques as Early Warning Indicators of Wetland Degradation: A Review *Toxicology & Water Quality.* 13:297-312.
Van Nes, E. H., Scheffer, M., van den Berg, M. S., and Coops, H. 2002. Dominance of charophytes in eutrophic shallow lakes—when should we expect it to be an alternative stable state? *Aquat. Bot.* 72:275-296.
Wei, A., Chow-Fraser, P., and Albert, D. 2004. Influence of shoreline features on fish distribution in the Laurentian Great Lakes. *Can. J. Fish. Aquat. Sci.* 61:1113-1123.
Wei, A., and Chow-Fraser, P. 2005. Untangling the confounding effects of urbanization and high water level on the cover of emergent vegetation in Cootes Paradise Marsh, a degraded coastal wetland of Lake Ontario. *Hydrobiologia* 544:1-9.
_____. 2006. Synergistic impact of water level fluctuation and invasions of Glyceria on Typha in a freshwater marsh of Lake Ontario. *Aquat. Bot.* 84:63-69.
_____. 2007. Development of an index to account for degree of exposure and wetland geomorphology on submerged aquatic vegetation of Lake Huron coastal wetlands.
Wilcox, D., Meeker, J. E., Hudson, P. L., Armitage, B. J., Black, M. G., and Uzarski, D. G. 2002. Hydrologic variability and the application of index of biotic integrity metrics to wetlands: a Great Lakes evaluation. *22* 3:588-615.
Zhang, J., Jorgensen, S. E., Beklioglu, M., and Ince, O. 2003. Hysteresis in vegetation shift— Lake Morgan prognosis. *Ecological Modeling* 164:227-238.

Table 1.1: Summary of environmental variables for the 154 wetland used for the CCA, where Turb = Turbidity, ISS = Inorganic Suspended Solids, OSS = Organic Suspended Solids, Chl a = Chlorophyll, Cond = Conductivity, TP = Total Phosphorus, TAN = Total Ammonia Nitrogen, TNN = Total Nitrate Nitrogen. Total Plant taxa refers to all emergent, submergent and floating plants, and # exotic taxa refers to only floating and submergent taxa. See text for explanation of WQI and the WMI scores.

Lake		WQI score	WMI score	Turb (NTU)	ISS (mg/L)	OSS (mg/L)	Chl a (µg/L)	pH	COND (µS/cm)	TP (µg/L)	TAN (mg/L)	TNN (mg/L)	Total plant taxa	# exotic taxa
Ontario	**Mean**	**-0.30**	**1.92**	**12.75**	**19.44**	**16.05**	**16.37**		**470**	**121.20**	**0.08**	**0.31**	**14**	**1**
(n=41)	Median	-0.29	2.00	5.22	2.40	4.60	9.20	7.80	349	97.67	0.04	0.25	12	1
	Min	-2.31	1.00	1.00	0.01	0.10	0.50	6.85	91	18.40	0.00	0.00	1	0
	Max	1.28	2.78	110.7	339.00	331.45	95.3	8.86	1658	407.00	0.55	1.03	36	4
Erie	**Mean**	**-0.35**	**2.06**	**22.61**	**15.82**	**7.58**	**24.82**		**388**	**117.10**	**0.11**	**0.54**	**13**	**1**
(n=24)	Median	-0.01	2.19	5.04	4.94	4.37	5.39	7.84	289	57.13	0.06	0.22	12	1
	Min	-2.86	1.00	1.27	0.26	1.20	0.57	6.90	185	19.77	0.00	0.10	2	0
	Max	0.73	3.00	226.30	116.80	67.00	360.7	8.74	860	569.20	0.64	3.20	28	3
Huron	**Mean**	**0.85**	**2.81**	**5.58**	**4.11**	**2.34**	**2.21**		**155**	**37.70**	**0.02**	**0.27**	**9**	**0**
(n=14)	Median	0.69	3.00	2.54	2.50	1.85	1.09	8.20	139	25.82	0.01	0.23	8	0
	Min	-0.41	1.58	0.69	0.53	0.21	0.12	7.48	69	3.12	0.00	0.00	3	0
	Max	2.36	3.42	32.40	18.46	6.00	9.94	8.78	239	117.10	0.07	0.63	18	2
Michigan	**Mean**	**-0.06**	**2.48**	**8.77**	**5.43**	**4.37**	**11.49**		**357**	**52.30**	**0.05**	**0.69**	**14**	**1**
(n=5)	Median	-0.30	2.41	6.17	5.93	5.40	4.78	8.27	361	52.30	0.05	0.87	16	1
	Min	-0.55	2.32	1.66	0.01	2.07	1.90	7.50	254	45.84	0.02	0.07	8	1
	Max	0.91	2.75	21.30	13.68	5.78	43.48	8.31	408	61.41	0.09	1.37	17	2
Georgian	**Mean**	**1.37**	**3.47**	**2.87**	**2.11**	**2.59**	**2.28**		**126**	**19.90**	**0.01**	**0.21**	**23**	**0**
Bay	Median	1.43	3.52	1.92	0.50	2.00	1.50	8.06	112	18.69	0.01	0.23	25	0
(n=49)	Min	-0.65	1.56	0.53	0.01	0.11	0.08	7.38	18	1.23	0.00	0.00	5	0
	Max	2.79	4.10	23.00	16.13	16.60	9.05	8.87	439	63.50	0.10	0.57	38	2
Superior	**Mean**	**0.53**	**2.82**	**10.75**	**5.33**	**3.82**	**4.00**		**141**	**42.30**	**0.03**	**0.29**	**16**	**0**
(n=21)	Median	0.67	3.00	6.12	2.39	3.60	3.98	7.61	134	39.50	0.02	0.15	17	0
	Min	-0.55	1.50	2.72	0.02	0.80	0.14	7.03	56	17.90	0.00	0.00	2	0
	Max	2.13	3.38	43.00	18.80	6.50	11.36	8.52	267	76.80	0.11	0.93	31	0

Table 1.2: Summary of U and T values for all taxa included in this study, organized according to habit type (emergent, floating and submergent). Common names and species codes are also included for convenience. U value indicates the tolerance of a species to degradation (1= very tolerant, 5= very intolerant) and T value indicates the niche breadth (1=broad niche, 3= narrow niche). % occurrence indicates the percentage of wetlands (n=176) in which the species in question occurred. * denotes that the species is not native to North America. Some species may be found in more than one group (e.g. emergent and floating) depending on the season.

Code	Taxon	Common name	U value	T value	% occurrence
Emergent					
ELAC	*Eleocharis acicularis*	needle spike rush	4	3	9.1
ELSM	*Eleocharis smallii*	marsh spike rush	4	2	32.9
EQFL	*Equisetum fluviatile*	water horsetail	4	2	6.8
ERAQ	*Eriocaulon aquaticum*	pipewort	5	3	17.6
LYSA	*Lythrum salicaria*	purple loosestrife*	1	1	21.6
PLAM	*Polygonum amphibium*	water smartweed	1	1	8.0
PLSP	*Polygonum sp.*	smartweed	1	1	4.5
POCO	*Pontederia cordata*	pickerelweed	3	2	48.3
SGCU	*Sagittaria cuneata*	small arrowhead	3	1	9.7
SGLA	*Sagittaria latifolia*	broad arrowhead	2	1	33.6
SGSP	*Sagittaria sp.*	Arrowhead species	2	1	6.8
SCAC	*Scirpus acutus*	hardstem bulrush	4	2	30
SCAM	*Scirpus americanus*	three-square bulrush	5	3	5.1
SCSP	*Scirpus sp.*	bulrush	4	1	31.8
SCVA	*Scirpus validus*	softstem bulrush	4	1	21.6
SPAD	*Sparganium androcladum*	branched burreed	4	3	2.3
SPAN	*Sparganium angustifolium*	narrow-leaf burreed	5	1	1.7
SPCL	*Sparganium chlorocarpum*	Green-fruit burreed	2	2	2.3
SPEM	*Sparganium emersum*	unbranched burreed	1	2	2.5
SPEU	*Sparganium eurycarpum*	giant burreed	3	2	10.8
SPSP	*Sparganium sp.*	burreed	2	2	15.3
TYAN	*Typha angustifolia*	narrow-leaf cattail*	1	1	21.0
TYLA	*Typha latifolia*	broadleaf cattail	3	2	16.5
TYSP	*Typha sp.*	cattail	1	1	23.3
TYXG	*Typha x glauca*	hybrid cattail*	1	2	7.4
UTCO	*Utricularia cornuta*	horned bladderwort	5	3	1.7
Floating					
BRSC	*Brasenia schreberi*	water shield	4	1	21

EICR	*Eichhornia crassipes*	water hyacinth*	1	1	0.6
HYMO	*Hydrocharis morsus-ranae*	frogbit*	1	2	11.4
LEMI	*Lemna minor*	lesser duckweed	1	1	11.4
LETR	*Lemna trisulca*	ivy duckweed	2	2	7.4
NELU	*Nelumbo lutea*	American lotus	1	1	1.2
NUAD	*Nuphar advena*	spatterdock	1	3	4.5
NUVA	*Nuphar variegata*	common yellow pond lily	2	1	56.7
NYOD	*Nymphaea odorata*	fragrant water lily (white)	2	1	66.5
NMCO	*Nymphoides cordata*	little floating hearts	5	3	2.8
PIST	*Pistia stratiotes*	water lettuce*	1	1	0.6
PONA	*Potamogeton natans*	broad-leaved pondweed	2	1	30.7
SPFL	*Sparganium fluctuans*	floating burreed	4	2	17.6
SPIR	*Spirodela polyrhiza*	greater duckweed	1	1	5.1
TRNA	*Trapa natans*	water chestnut*	1	1	0.6
WOLF	*Wolffia sp.*	watermeal*	1	2	1.7
Submergent					
BIBE	*Bidens beckii*	Beck's marsh marigold	4	2	22.7
CABO	*Cabomba*	fanwort	1	1	4.5
CASP	*Callitriche sp.*	water starwort	4	2	10.2
CEDE	*Ceratophyllum demersum*	coontail	1	1	45.5
CHSP	*Chara sp.*	muskgrass	3	2	55.1
ELCA	*Elodea canadensis*	Canadian waterweed	2	1	63.6
HIVU	*Hippuris vulgaris*	mare's tail	3	3	1.7
ISSP	*Isoetes sp.*	quillwort	4	3	12.5
LODO	*Lobelia dortmanna*	water lobelia	5	2	6.3
MYAL	*Myriophyllum alterniflorum*	alternate water-milfoil	5	3	7.4
MYFA	*Myriophyllum farwellii*	Farwell's water-milfoil	3	1	0.6
MYHE	*Myriophyllum heterophyllum*	two-leaf water-milfoil	3	2	8.0
MYSI	*Myriophyllum sibiricum*	common water-milfoil	3	2	35.8
MYSC	*Myriophyllum spicatum*	Eurasian water-milfoil*	1	1	30.7
MYTE	*Myriophyllum tenellum*	slender water-milfoil	4	3	8.5
MYVE	*Myriophyllum verticillatum*	whorled water-milfoil	4	1	0.6

MYSP	*Myriophyllum sp.*	Water-milfoil	1	1	30.1
NAFL	*Najas flexilis*	slender water nymph	3	2	51.7
NEAQ	*Neobeckia aquatica*	north American Lake-Cress	5	3	1.1
NISP	*Nitella sp.*	stonewort	3	1	13.1
POAM	*Potamogeton amplifolius*	large-leaved pondweed	4	2	25.0
POCR	*Potamogeton crispus*	curly-leaf pondweed*	1	1	25.6
POEP	*Potamogeton epiphydrus*	ribbon-leaf pondweed	4	3	10.8
POFO	*Potamogeton foliosus*	leafy pondweed	2	1	0.6
POFR	*Potamogeton friesii*	Fries' Pondweed	2	1	1.1
POGR	*Potamogeton gramineus*	variable pondweed	4	2	29.5
POIL	*Potamogeton illinoensis*	Illinois pondweed	3	2	8.0
POOB	*Potamogeton obtusifolius*	bluntleaf pondweed	2	1	0.6
PO SLEN	*Potamogeton pusillus*	"slender" pondweed	2	1	2.3
PORI	*Potamogeton richardsonii*	clasping-leaved pondweed	3	2	64.8
PORO	*Potamogeton robbinsii*	fern-leaf pondweed	4	2	25.0
POSP	*Potamogeton sp.*	pondweed	1	2	21.0
POSR	*Potamogeton spirillus*	northern snailseed pondweed	5	2	14.2
POVA	*Potamogeton vaseyi*	Vaseyi pondweed	2	1	0.6
POZO	*Potamogeton zosteriformis*	flat-stemmed pondweed	3	1	38.1
RALO	*Ranunculus longirostris*	buttercup, crowfoot	2	1	16.5
RASP	*Ranunculus sp*	crowfoot	2	1	1.1
SGGR	*Sagittaria graminea*	grassy arrowhead	4	3	5.7
SCSU	*Scirpus subterminalis*	water bulrush	5	2	13.6
SPON	Fresh water sponges	sponges	5	3	9.7
STPE	*Stuckenia pectinata*	sago pondweed	1	1	37.5
STVA	*Stuckenia vaginata*	sheathed pondweed	2	1	0.6
UTGE	*Utricularia geminiscapa*	hidden fruit bladderwort	5	3	1.1
UTGI	*Utricularia gibba*	humped bladderwort	5	2	1.1
UTIN	*Utricularia intermedia*	flatleaved bladderwort	3	2	5.1
UTMI	*Utricularia minor*	lesser bladderwort	5	2	1.7
UTPU	*Utricularia purpurea*	purple bladderwort	5	2	1.7
UTSP	*Utricularia sp.*	bladderwort	3	2	4.0
UTVU	*Utricularia vulgaris*	common bladderwort	3	2	30.0
VAAM	*Vallisneria americana*	tape grass, eel grass	3	1	64.2
ZIPA	*Zizania sp.*	wild rice	4	2	30.1
ZODU	*Zosterella dubia*	water stargrass	2	2	5.7

Table 1.3: Range of WMI scores associated with the different categories

Category	Range
Excellent	>3.51
Very good	3.01-3.50
Good	2.51-3.00
Moderately degraded	2.01-2.50
Very degraded	1.51-2.00
Highly degraded	1.00-1.50

Table 1.4: Summary of percent composition of submersed aquatic plant species found at 8 stations in Sturgeon Bay in 1988 and 2004. No significant difference between Shannon-Weiner diversity index for 1988 and 2004 (t-test, P= 0.52)

	Percent composition of species in 1988								Percent composition of species in 2004							
Taxon and variable	**532**	**534**	**536**	**537**	**539**	**541**	**542**	**543**	**532**	**534**	**536**	**537**	**539**	**541**	**542**	**543**
Ceratophyllum demersum	5	15	5	5	5	5	5	5	0	1	0	1	15	1	10	0
Chara sp.	0	0	0	0	0	0	0	10	0	0	0	0	0	0	0	1
Elodea canadensis	10	30	25	5	25	0	65	30	1	10	1	30	35	1	10	0
Freshwater Sponge	0	0	0	0	0	0	0	0	0	0	0	0	10	0	10	0
Zosterella dubia	1	1	0	1	0	1	1	1	0	1	30	10	0	0	30	1
Bidens beckii	1	0	1	0	0	0	1	1	1	0	25	0	10	0	1	0
Myriophyllum sibiricum	0	1	1	1	1	1	5	1	30	25	30	25	25	0	65	25
Myriophyllum spicatum	60	30	30	75	65	5	10	35	0	0	0	10	0	0	0	0
Najas flexilis	1	1	1	1	1	20	0	5	0	1	35	0	0	0	0	0
Nuphar variegata	0	0	0	1	0	0	0	1	0	0	0	0	0	0	0	0
Potamogeton amplifolius	0	0	0	0	0	0	0	0	1	10	1	1	1	0	0	0
Potamogeton crispus	0	1	1	0	0	1	1	0	0	0	0	0	0	0	0	0
Potamogeton foliosus	1	1	0	1	1	0	0	0	0	0	0	0	0	0	0	1
Potamogeton praelongus	0	1	1	0	1	1	1	0	0	0	1	0	1	0	10	0
Potamogeton richardsonii	1	10	1	1	5	1	5	5	1	10	1	1	1	1	10	1
Potamogeton robbinsii	0	0	1	0	0	0	0	0	0	0	0	1	0	0	0	0
Potamogeton zosteriformis	0	0	0	0	0	0	0	0	1	0	1	1	15	1	15	0
Ranunculus longirostris	0	1	0	0	0	0	0	1	0	0	0	0	0	0	0	0
Vallisneria americana	15	30	20	5	0	60	0	15	55	55	0	65	0	0	35	30
Shannon-Weiner diversity index	0.52	0.83	0.65	0.41	0.48	0.59	0.50	0.84	0.40	0.65	0.72	0.64	0.82	0.08	1.08	0.39
Number of species	9	12	11	10	8	9	9	12	7	8	9	10	9	4	10	6

Table 1.5: Summary of the WMI scores associated with eight sites sampled for plants in Sturgeon Bay during 1988 and 2004. Distance from shore and proximity to sewage outflow pipe were determined by GIS (ArcView 3.2). Rank of proximity to sewage outflow pipe was given a value of 1 to 8 where the sampling site closest to the sewage outflow was given a value of 1 and the furthest was given a value of 8. Mean WMI scores from 1988 were significantly lower than 2004 ($P<0.0021$).

Site	WMI 1988	WMI 2004	Change in WMI	Distance from Shore (m)	Rank of proximity to sewage outflow pipe
532	2.58	3.27	0.69	293.06	4
534	2.29	2.77	0.48	400.55	1
536	2.80	3.07	0.27	210.27	5
537	2.13	2.80	0.68	784.38	3
539	2.40	3.50	1.10	570.94	2
541	2.33	2.40	0.07	1227.55	6
542	2.42	3.20	0.78	663.79	7
543	2.40	2.70	0.30	551.11	8

Table 1.6: Location and description of human impacts on wetlands of the two Canadian National parks.

National park	Wetland	Date sampled	Latitude	Longitude	Type of impact
PPNP	Sanctuary Pond	June 20, 2005	41.98032	-82.54189	Sewage, agricultural run-off, common carp (*Cyprinus carpio)*
PPNP	West Cranberry Pond	June 21, 2005	41.97453	-82.51620	Common carp (*Cyprinus carpio)*
PPNP	East Cranberry Pond	June 21, 2005	41.97153	-82.50759	Common carp(*Cyprinus carpio)*
PPNP	Big Pond 1	June 20, 2005	41.96565	-82.52061	Common carp (*Cyprinus carpio)*
FFNMP	Cove North	July 7, 2005	45.31340	-81.76227	No human impact
FFNMP	Boat Passage	July 6, 2005	45.28953	-81.71899	Boat channel
FFNMP	Hay Bay 1	July 4, 2005	45.24089	-81.68385	Public beach, high cottage density
FFNMP	Hay Bay 2	July 7, 2005	45.23341	-81.69424	Low cottage density

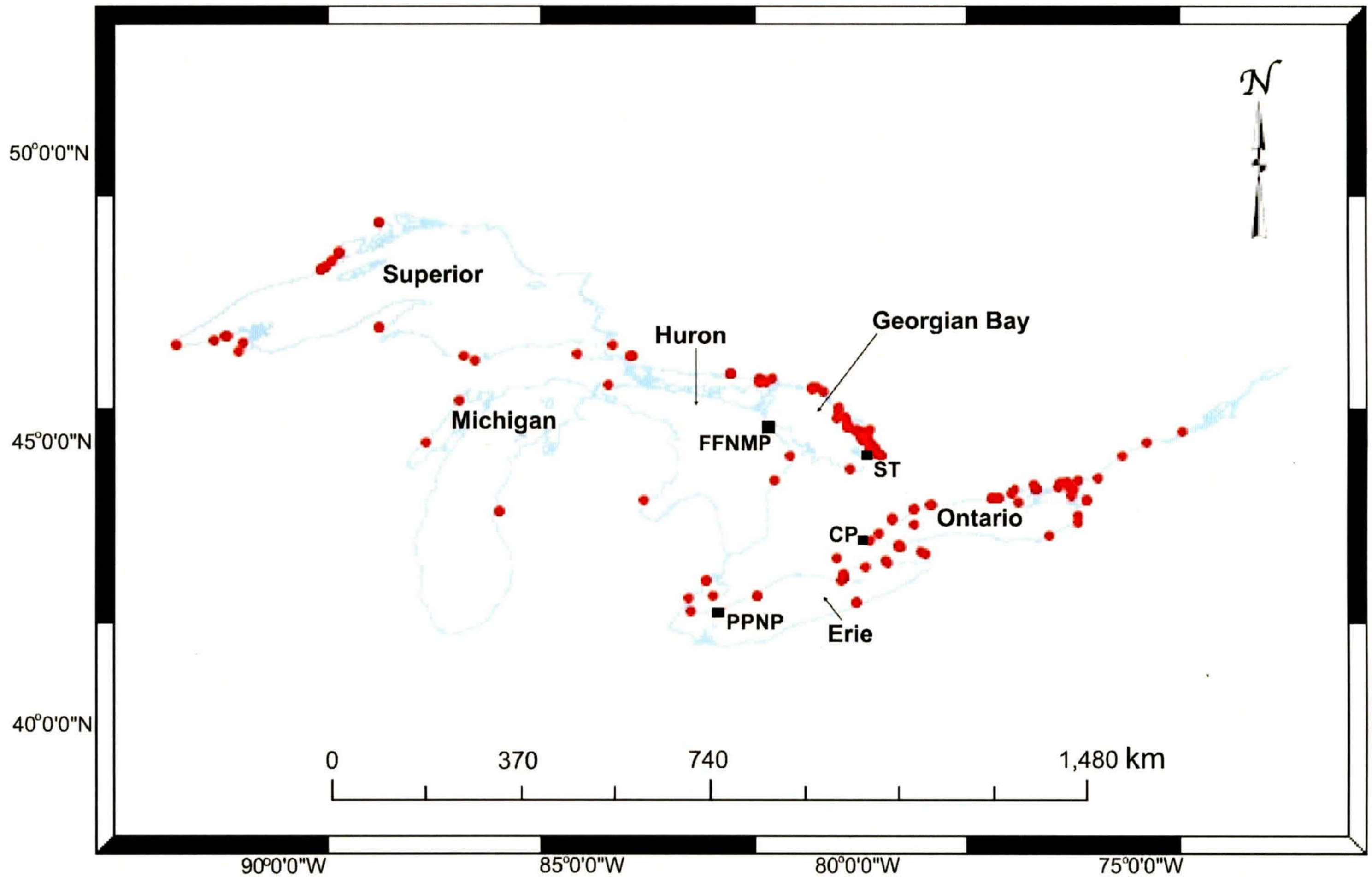


Figure 1.1: Location of 176 wetland years used in the application of the WMI. Location of the four study sites used for validation of the WMI are indicated by square symbols. FFNMP=Fathom Five National Marine Park, ST=Sturgeon Bay, CP=Cootes Paradise, and PPNP=Point Pelee National Park.

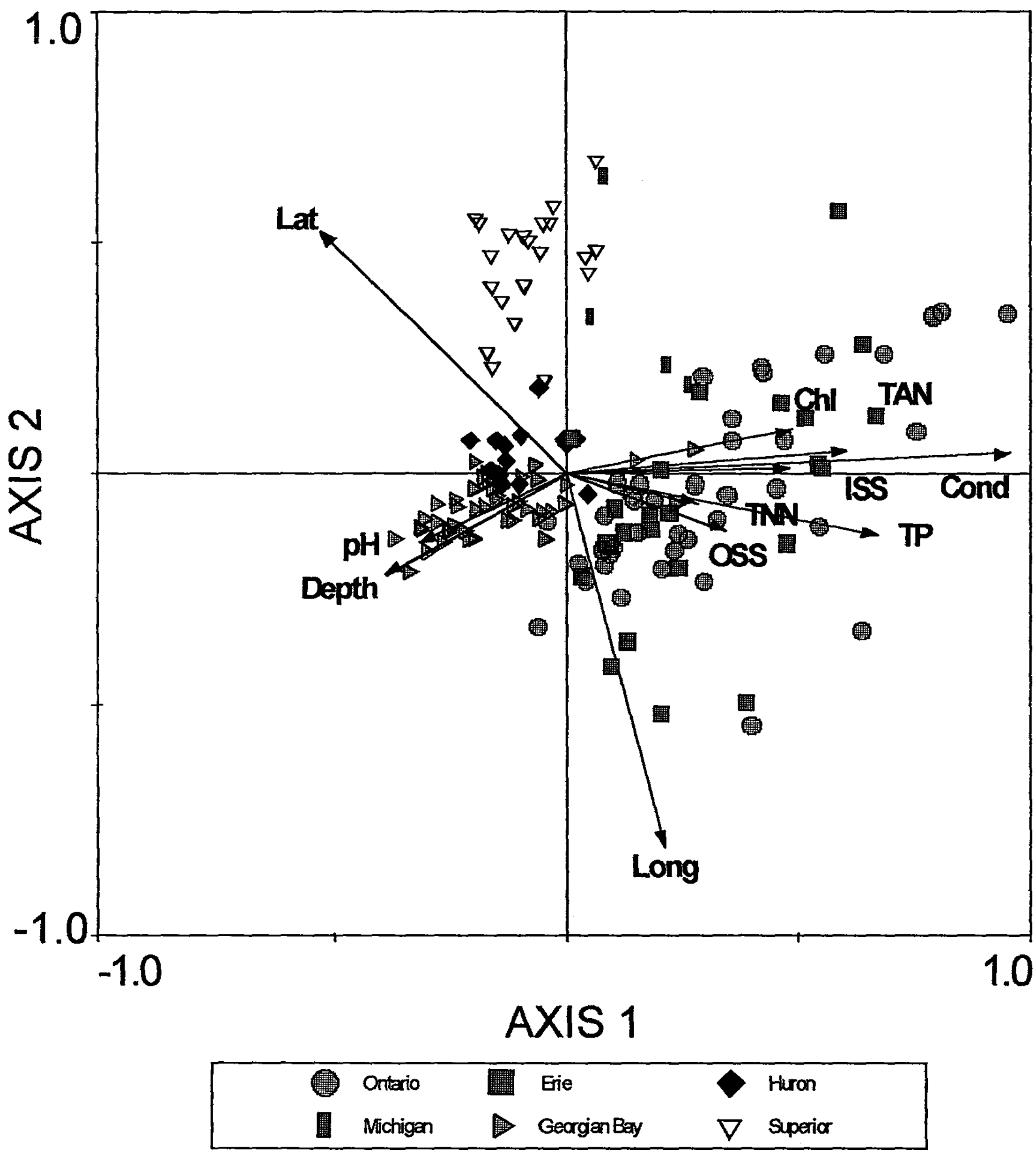


Figure 1.2: Bi-plot of CCA Axis 1 versus CCA Axis 2. Vectors for the 11 environmental variables are shown (lines with arrows emanating from the origin). The strength of the correlation of the environmental variable with the axis is a direct function of the length of the vector and how close it is to the axis. Points represent the 154 wetland years used in the CCA. Wetlands were grouped after the CCA by lake, for ease of interpretation. Wetlands on the right side of the plot have higher nutrient and turbidity levels.

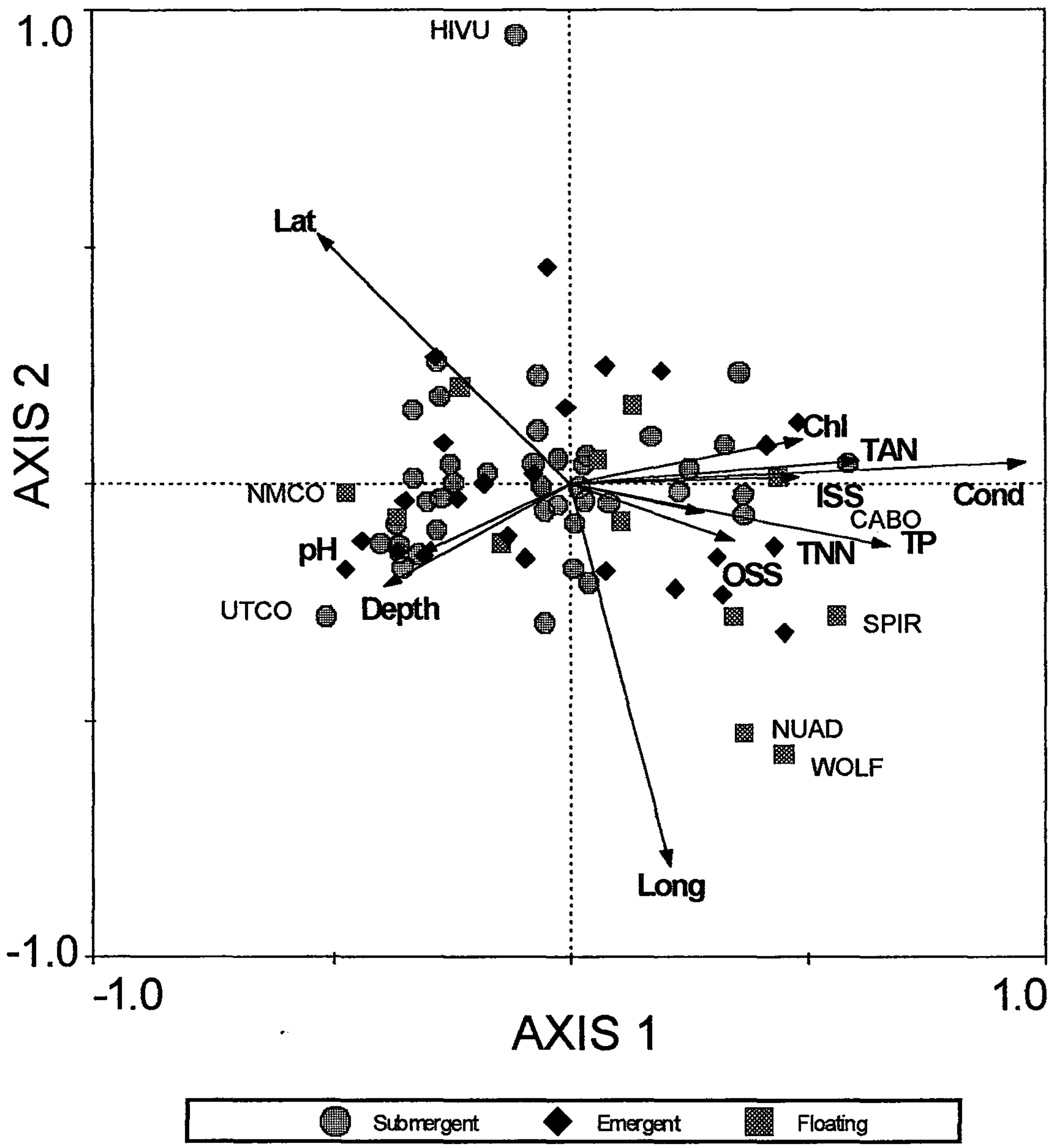
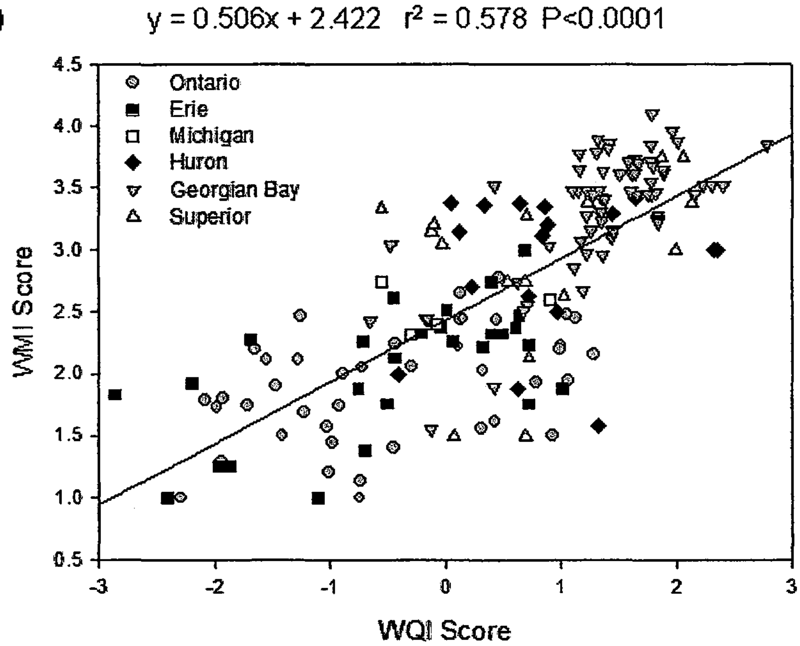


Figure 1.3: Bi-plot of CCA Axis 1 versus CCA Axis 2. Vectors for the 11 environmental variables are shown (lines with arrows emanating from the origin). Some species centroids are identified with a 4-letter code (see Table 2 for explanation of codes). Species were grouped by habitat. Species found on the left of the plot are intolerant of degradation.

a)



b)

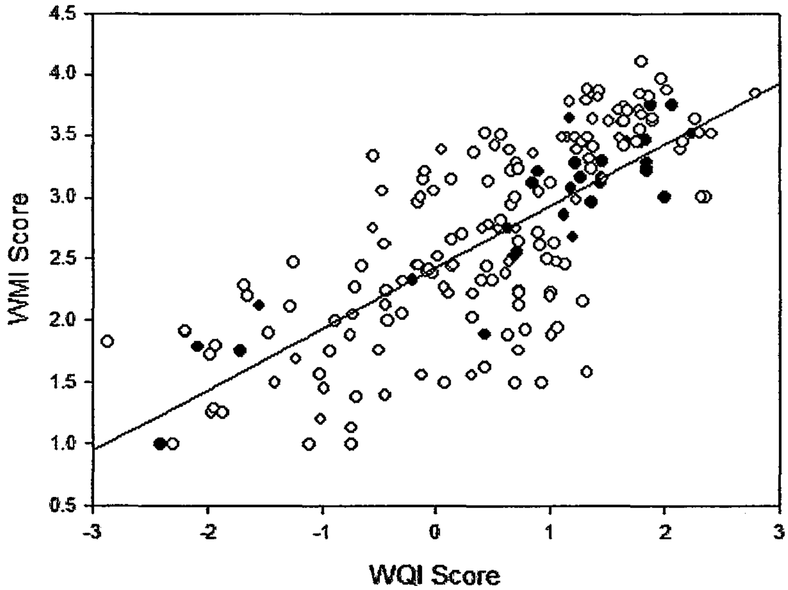
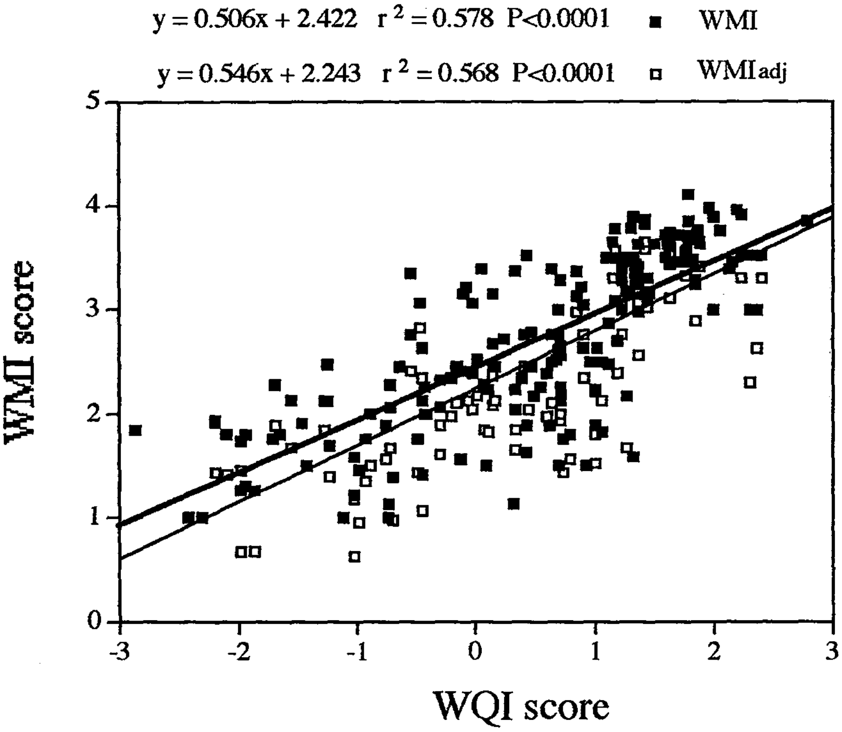


Figure 1.4: Relationship between the WMI score and WQI score for 176 wetland-years a) wetlands grouped by lake, and b) open circles represent 154 wetland-years used for both the development of the WMI and for application of the WMI, closed circles represent the 22 wetland-years that were used only for the application of the WMI

a)



b)

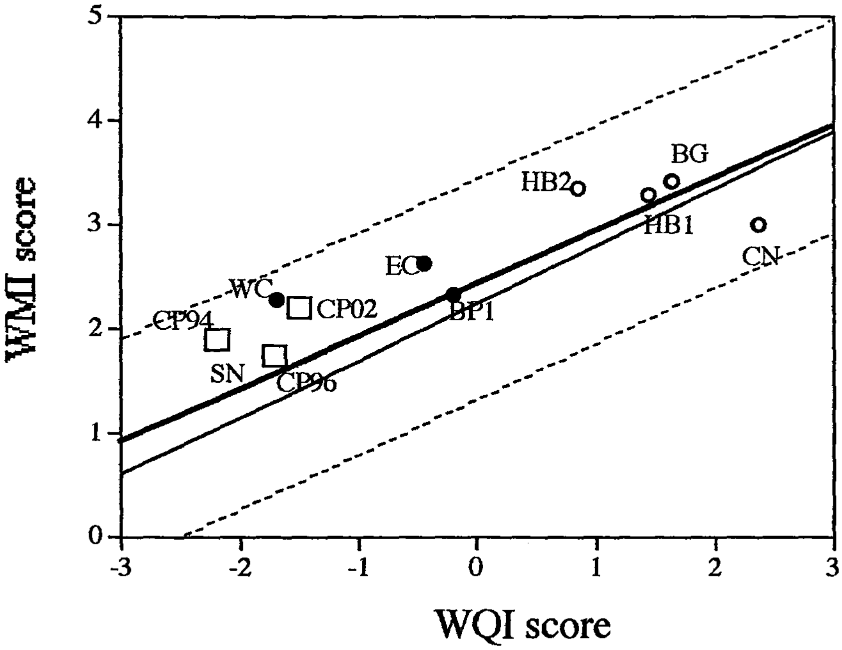


Figure 1.5: a) Plot of the WMI (closed squares) and the WMIadj (open squares) vs. WQI score for 176 wetland-years. b) Plot of the WMI vs. WQI score corresponding to Cootes Paradise Marsh (open squares), and wetlands of Point Pelee National Park (closed circles)

and Fathom Five National Marine Park (open circles). The regression lines correspond directly to those in Figure 1.5a. Dashed lines represent the 95% confidence intervals.

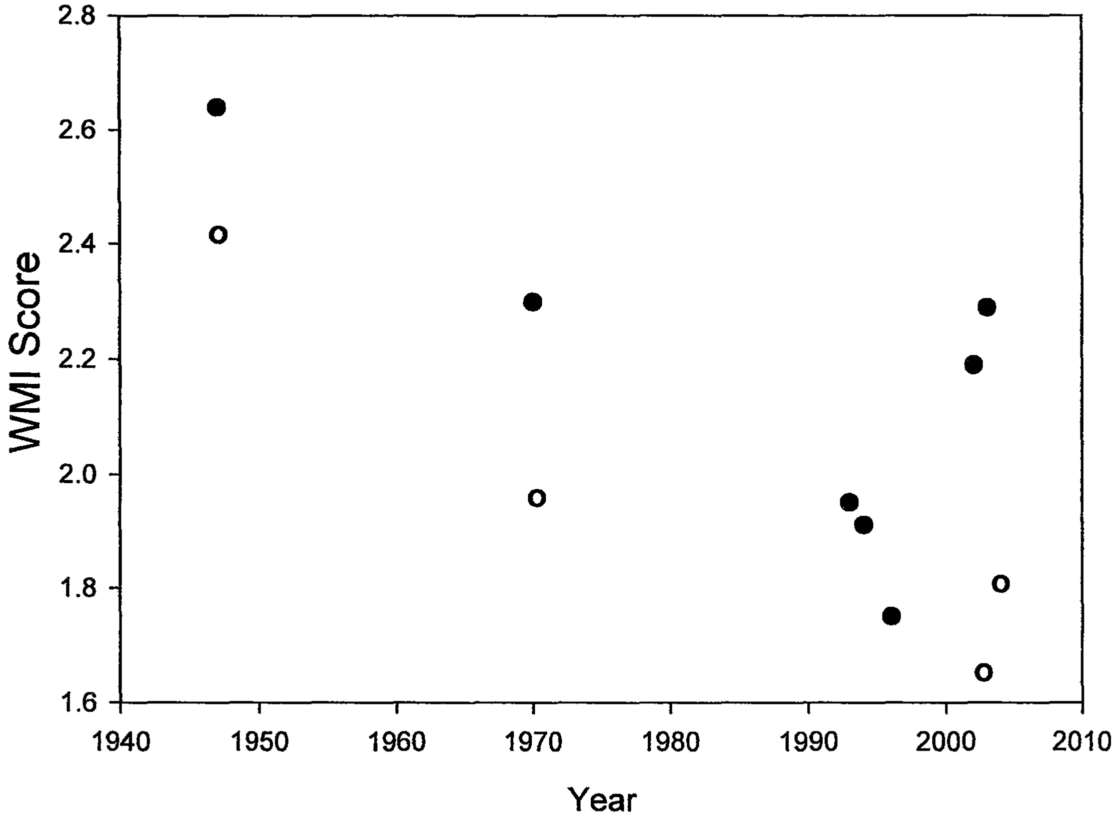


Figure 1.6: Relationship between the WMI (solid symbol) and the WMIadj (open symbol) for Cootes Paradise Marsh from 1946 to 2003. The WMI and WMIadj scores during the 1990s had the same value.

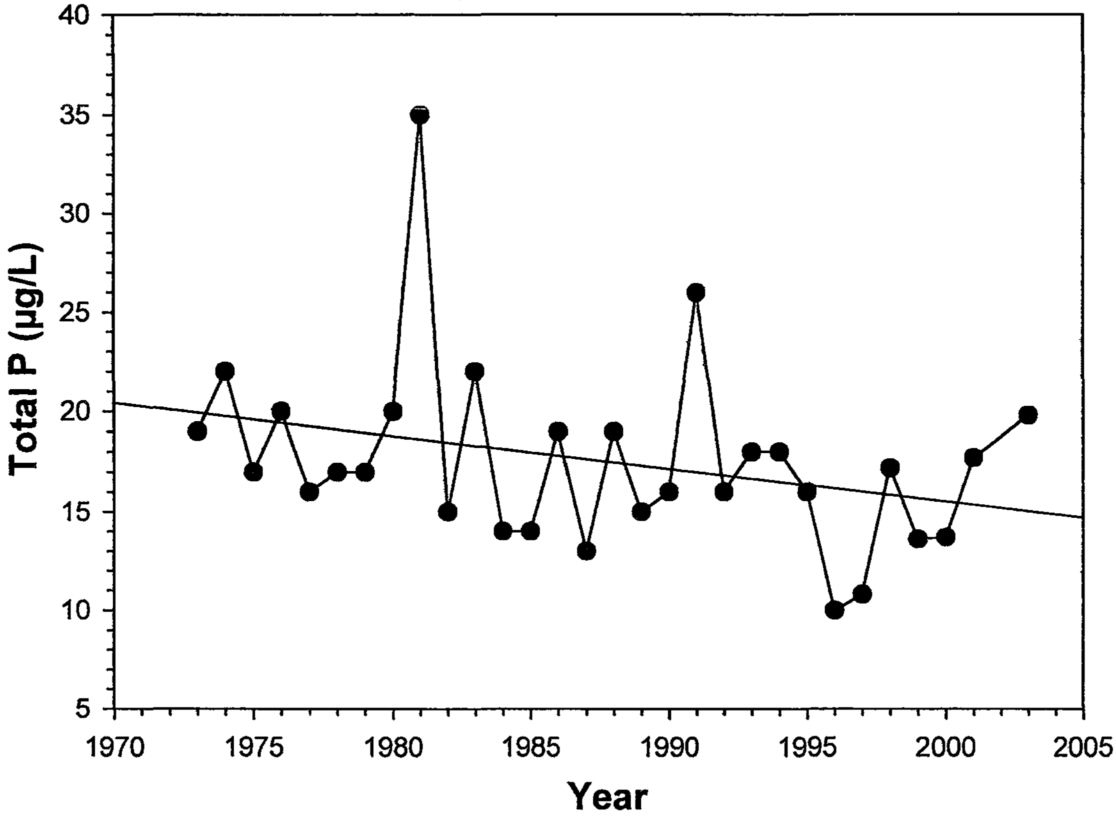


Figure 1.7: Change in TP concentrations in Sturgeon Bay from 1972 to 2003. There was a decline in nutrient concentrations through time as indicated by a linear regression analysis (P=0.06).

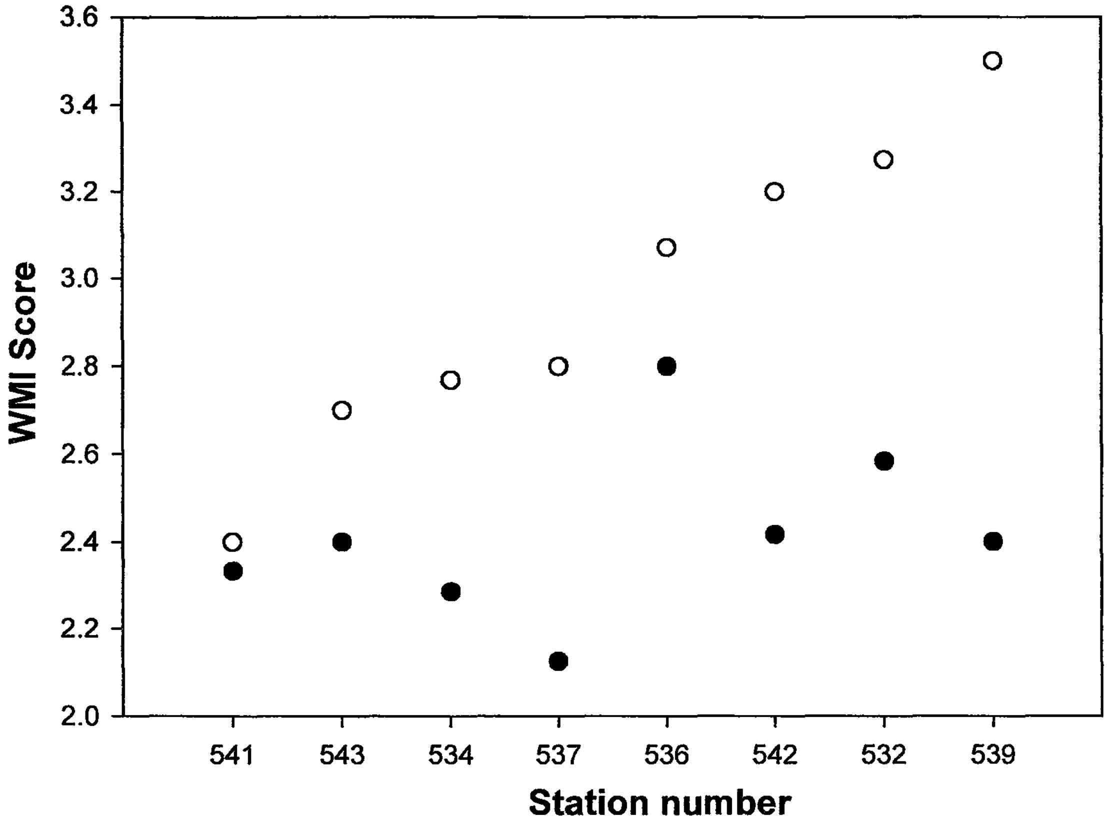


Figure 1.8: Relationship between the WMI scores calculated for 8 stations in Sturgeon Bay between 1988 (closed circles), prior to RAP delisting and 2004 (open circles), following RAP delisting. There were significant differences between the two time periods (Paired t-test; P = 0.0012)

Appendix 1.1: List of 154 wetland years and their locations (latitude and longitude) used for the development of the WMI.

Wetland	Year	Lake	WMI score	WMI adj score	Latitude	Longitude
Big Creek (Teeterville)	1996	Erie	2.13	2.13	45.95550	-80.44719
East Cranberry	2005	Erie	2.62	2.34	41.97153	-82.50759
Grand River	1998	Erie	1.25	0.67	42.88390	-79.57220
Grand River	2001	Erie	1.25	0.67	42.90000	-79.60000
Holiday Conservation	1996	Erie	1.83	1.83	42.03335	-83.05000
Long Point Big Rice	2001	Erie	2.38	2.03	42.58930	-80.33550
Long Point Inner Bay	2001	Erie	2.38	1.97	42.59650	-80.34180
Long Point Inner Channel	2001	Erie	3.00	2.59	42.59130	-80.33550
Long Point Little Rice	2000	Erie	2.22	1.83	42.58930	-80.33550
Long Point Prov Park	1998	Erie	2.24	1.93	42.58333	-80.38333
Presque Isle	2000	Erie	2.52	2.17	42.15900	-80.09850
Redhead Pond	2005	Erie	2.27	1.84	41.95378	-82.50657
Rondeau Bay	1998	Erie	2.33	2.33	42.30070	-81.85530
Rondeau Bay	2001	Erie	2.75	2.45	42.28800	-81.86700
Sanctuary Pond	2005	Erie	1.92	1.42	41.98032	-82.54189
Selkirk Prov Park	1998	Erie	1.38	0.97	42.81667	-79.95000
Turkey Creek	1996	Erie	1.88	1.56	42.23556	-83.08528
Turkey Point	1998	Erie	2.17	2.17	42.66860	-80.35320
Turkey Point	2002	Erie	2.48	2.09	42.63359	-80.34170
West Cranberry	2005	Erie	2.28	1.87	41.97453	-82.51620
Cormican Bay	2003	Georgian Bay	3.82	3.82	45.40765	-80.31288
Cow Island	2005	Georgian Bay	3.78	2.78	46.09859	-81.81942
David's Bay	2004	Georgian Bay	3.48	3.48	45.04750	-80.00380
Dead Horse	2005	Georgian Bay	3.23	3.23	46.10463	-81.60802
Dogfish Bay	2005	Georgian Bay	3.28	3.05	46.08091	-81.73593
French River Main	2005	Georgian Bay	3.73	3.52	45.96796	-80.88779
Ganyon Bay	2005	Georgian Bay	3.86	3.64	44.92052	-79.81763
Garden Channel	2003	Georgian Bay	3.61	3.61	45.18628	-80.12147
Gooseneck	2004	Georgian Bay	3.15	3.15	45.20688	-80.10749
Green Island	2003	Georgian Bay	3.04	2.76	44.78833	-79.74403
Green Island	2004	Georgian Bay	3.40	3.16	44.78862	-79.74900
Herman's Bay	2004	Georgian Bay	3.62	3.62	45.08638	-79.99758
Herman's Bay	2005	Georgian Bay	3.71	3.50	45.21824	-79.86969
Hockey Stick Bay	2005	Georgian Bay	3.81	3.58	44.94461	-79.86297

Hole in the Wall	2005	Georgian Bay	3.63	3.63	45.52182	-80.43831
Ingersoll Bay	2005	Georgian Bay	3.84	3.84	45.28132	-80.25588
Jumbo Bay	2005	Georgian Bay	3.71	3.71	46.05244	-81.81858
Key River	2003	Georgian Bay	3.22	2.99	45.88742	-80.67858
Lily Pond	2005	Georgian Bay	3.05	2.82	44.87037	-79.81478
Longuissa Bay	2003	Georgian Bay	3.51	3.30	44.96723	-79.89152
Matchedash Bay	1998	Georgian Bay	1.56	1.56	44.73333	-79.66667
Matchedash Bay	2002	Georgian Bay	2.44	2.44	44.73353	-79.66683
Matchedash Bay	2003	Georgian Bay	2.45	2.10	44.75520	-79.69648
Moon River Bay	2003	Georgian Bay	3.63	3.36	45.12053	-79.97500
Moon River Falls	2003	Georgian Bay	3.52	3.52	45.10733	-79.92995
Moose Bay	2003	Georgian Bay	3.31	3.31	45.07210	-80.04958
Moreau Bay	2003	Georgian Bay	3.70	3.70	45.01092	-79.94572
Musky Bay	2003	Georgian Bay	3.48	3.30	44.81040	-79.78265
Musky Bay	2004	Georgian Bay	3.48	3.30	44.81040	-79.78265
Ni Bay	2005	Georgian Bay	3.44	3.44	45.40924	-80.45599
North Bay	2005	Georgian Bay	3.52	3.52	44.89638	-79.79377
Oak Bay	2003	Georgian Bay	2.86	2.86	44.79630	-79.73158
Ojibway Bay	2005	Georgian Bay	3.67	3.44	44.88758	-79.85585
Otter Creek	2005	Georgian Bay	3.77	3.56	45.95403	-80.82421
Port Rawson	2003	Georgian Bay	3.44	3.44	45.19512	-80.02350
Quarry Island	2003	Georgian Bay	3.48	3.48	44.83400	-79.80968
Quarry Island	2004	Georgian Bay	3.48	3.48	44.83217	-79.80550
Sandy Island	2003	Georgian Bay	3.87	3.87	45.26865	-80.25065
Sandy Island West	2005	Georgian Bay	3.64	3.40	45.27659	-80.26755
Sturgeon Central	2003	Georgian Bay	3.42	3.23	45.61782	-80.43260
Tadenac Bay	2004	Georgian Bay	3.88	3.88	45.13742	-79.99287
Tadenac Bay 1	2005	Georgian Bay	4.10	4.10	45.03444	-79.99145
Tadenac Bay 2	2005	Georgian Bay	3.96	3.86	45.03916	-79.98792
Tadenac Lake	2005	Georgian Bay	3.84	3.84	45.03437	-79.95509
Treasure Bay	2005	Georgian Bay	3.55	3.32	44.86854	-79.86049
Waldon's Pond	2005	Georgian Bay	3.62	3.62	45.92294	-80.87577
Wardrope Island	2005	Georgian Bay	3.44	3.46	46.05486	-81.71651
West Bay	2003	Georgian Bay	3.50	3.50	45.42228	-80.30727
Baie du Dore	1998	Huron	1.58	1.58	44.33670	-81.55570
Boat Passage	2005	Huron	3.42	3.10	45.28953	-81.71899
Collingwood Harbour	1998	Huron	2.00	2.00	44.50920	-80.23260
Cove Island North	2005	Huron	3.00	2.62	45.31340	-81.76227
Echo Bay	1998	Huron	1.88	1.88	46.49453	-84.07597
Echo Bay	2000	Huron	3.38	3.38	46.49453	-84.07597
Echo Bay	2002	Huron	3.38	3.38	46.49460	-84.05500
Hay Bay 2	2005	Huron	3.35	2.97	45.23341	-81.69424
Mismer	2000	Huron	3.14	3.14	46.00510	-84.46060
Oliphant Bay	1998	Huron	2.64	2.64	44.73131	-81.28203

Russell Island West	2005	Huron	3.00	2.29	45.26458	-81.70412
Spanish River	1998	Huron	3.36	3.36	46.18339	-82.35000
Spanish River	2000	Huron	2.70	2.70	46.17845	-82.34585
Spanish River	2002	Huron	2.50	2.17	46.18339	-82.31691
Lake St. Clair	1999	Lake St. Clair	1.76	1.43	44.58333	-82.76667
Lake St. Clair	2000	Lake St. Clair	1.76	1.43	44.58333	-82.76667
Tremblay Beach	1998	Lake St. Clair	1.00	1.00	42.30000	-82.65000
Pentwater Marsh	2000	Michigan	2.32	1.87	43.76280	-86.40780
Pentwater Marsh	2001	Michigan	2.32	1.87	43.76280	-86.40780
Peshtigo	2001	Michigan	2.61	2.33	44.98400	-87.66070
Portage Creek	2001	Michigan	2.75	2.40	45.70620	-87.08000
Wigwam Bay	2001	Michigan	2.41	2.13	43.97020	-83.85430
Buckhorn	2001	Niagara	2.27	2.27	43.05630	-78.97120
Spicer Creek	2001	Niagara	1.88	1.52	43.02338	-78.89677
Bayfield Marsh	2000	Ontario	1.75	1.34	44.19758	-76.36500
Blessington Bay	2002	Ontario	2.44	2.07	44.16700	-77.33300
Bronte Creek	2002	Ontario	1.45	0.95	43.39340	-79.71546
Credit River	2002	Ontario	1.90	1.90	43.55007	-79.08358
Darlington	2001	Ontario	1.20	0.62	43.87300	-78.79700
Fifteen Mile Creek	2002	Ontario	1.73	1.44	43.16693	-79.31668
Frenchman's Bay	1998	Ontario	2.00	1.50	43.82240	-79.09490
Frenchman's Bay	2001	Ontario	2.06	1.59	43.81233	-79.09467
Goose Bay	2002	Ontario	2.22	1.82	44.35005	-75.86671
Grass Bay	2002	Ontario	2.46	2.46	44.15018	-76.26681
Grindstone Creek	2002	Ontario	1.00	1.00	43.28333	-79.88333
Hay Bay Marsh	1996	Ontario	2.23	2.23	44.16675	-76.93335
Hay Bay Marsh	2000	Ontario	2.45	2.11	44.16675	-76.93335
Hay Bay Marsh	2002	Ontario	2.44	2.04	44.16675	-76.93335
Humber River	1996	Ontario	1.80	1.80	43.64280	-79.48860
Humber River	2002	Ontario	1.50	1.50	43.61673	-79.48333
Johnstown Creek	1998	Ontario	1.69	1.38	44.73300	-76.46700
Jordan Harbour	1996	Ontario	1.80	1.80	43.17930	-79.37340
Jordan Harbour	2002	Ontario	1.29	1.29	43.15014	-79.38333
Little Cataraqui Creek	1998	Ontario	1.00	1.00	44.28110	-76.51630
Little Cataraqui Creek	2002	Ontario	2.11	1.84	44.21667	-76.55000
Little Sodus	2001	Ontario	2.03	1.65	43.33942	-76.69447
Madoma Creek	1998	Ontario	1.50	1.50	44.26667	-76.38333
Madoma Creek	2002	Ontario	2.23	1.99	44.26667	-76.38333
Mud Bay	2002	Ontario	2.05	1.66	44.06682	-76.31672
Muskellunge River	2002	Ontario	2.24	1.99	43.96682	-76.05010
Napanee River	1998	Ontario	1.40	1.05	44.23333	-76.98333
Perch River	2002	Ontario	2.66	2.35	43.98361	-76.06688

Presqu'ile Prov Park	1998	Ontario	1.81	1.81	44.00000	-77.73060
Presqu'ile Prov Park	2002	Ontario	2.78	2.44	44.00000	-77.73060
Salmon River	2002	Ontario	2.16	1.66	48.56667	-76.20004
Sandy Creek	2002	Ontario	2.48	2.11	43.70089	-76.19647
Sawguin Creek	1996	Ontario	1.62	1.62	44.10000	-77.38333
Second Marsh	1995	Ontario	2.47	2.11	43.87500	-78.81320
Weller's Bay	1998	Ontario	1.80	1.56	44.01679	-77.61670
Wellers Bay	2002	Ontario	2.20	1.79	44.01679	-77.61670
West Lake	1998	Ontario	1.11	1.11	43.93333	-72.28333
Pt. Mouillee	1998	St. Lawrence	1.13	1.13	45.16667	-74.36667
Upper Canada Bird Sanctuary	1998	St. Lawrence	2.40	2.40	44.98300	-75.00000
Willowbank Marsh	1998	St. Lawrence	1.57	1.16	44.31667	-76.21667
Au Train	2002	Superior	2.94	2.94	46.43334	-86.81681
Bark Bay	2000	Superior	3.13	3.13	46.85042	-91.19819
Chippewa Creek	1998	Superior	1.50	1.50	48.33870	-89.21570
Chippewa Park	2002	Superior	1.50	1.50	48.31700	-89.20000
Cloud Bay	2001	Superior	3.38	3.38	48.08120	-89.44370
Cloud Bay	2002	Superior	3.38	3.38	48.08280	-89.43720
Flag	2002	Superior	3.14	3.14	46.78667	-91.38778
Goulais River Oxbow	1998	Superior	2.25	2.25	46.71667	-84.41667
Hurkett Cove	1998	Superior	2.13	2.13	48.83300	-88.50000
Hurkett Cove	2002	Superior	3.21	3.21	48.83080	-88.49470
Laughing Whitefish	2002	Superior	3.23	3.01	46.51675	-87.01688
Lost Creek	2001	Superior	3.28	3.28	46.85861	-91.13583
Nemadji River	2002	Superior	2.96	2.96	46.68353	-92.03340
Pike River	2002	Superior	3.00	3.12	47.01676	-88.51679
Pine Bay	1998	Superior	3.05	3.05	48.03360	-89.52320
Pine Bay	2001	Superior	3.33	3.33	48.03330	-89.51950
Sioux River	2000	Superior	2.81	2.81	46.73430	-90.87790
Sturgeon Bay Slough	2002	Superior	3.00	3.00	47.00024	-88.48348
Sturgeon Bay Superior	1998	Superior	2.63	2.63	48.19020	-89.31160
Taquamenon River	2002	Superior	2.71	2.71	46.55010	-85.01691
West Fish Creek	2001	Superior	2.75	2.75	46.58420	-90.94610

Chapter 2

Non-random sampling and its role in habitat conservation: A comparison of three wetland macrophyte sampling protocols

Melanie V. Croft
And
Pat Chow-Fraser

McMaster University
Dept. of Biology, 1280 Main St. West
Hamilton, Ontario, Canada
L8S 4K1

Abstract

Aquatic macrophytes provide essential spawning and nursery habitat for fish, valuable food source for waterfowl, migratory birds and mammals, and contribute greatly to overall biodiversity of coastal marshes of the Laurentian Great Lakes. Two approaches have been used to survey the plant community in coastal wetlands, and these include the grid (GR) and transect (TR) (more common) methods, which rely on placement of grids or random transects at a site. These methods have been used to determine the average condition of species richness at different sites, but their suitability for surveying total species richness of a particular site has not yet been tested. In this paper, we compare the performance of these two established methods with that of the Stratified method (ST), which uses the sampler's judgment to guide them to different habitat zones during the macrophyte survey. We used the three protocols to compare species richness of six coastal wetlands of the Great Lakes, three pristine marshes in remote regions of eastern Georgian Bay (Lake Huron) and three degraded wetlands in populated areas of Lake Ontario, Canada. The greatest species richness was associated with the ST method, irrespective of the quality of wetlands. The ST method was also more efficient (highest number of species per unit time), and revealed the most number of unique (those found only in samples collected by that particular method) and uncommon species (those found in <5% of the quadrats) compared with the GR and TR methods. Despite these statistical differences, we found that sampling method did not significantly affect the performance of a recently developed index of wetland quality, the Wetland Macrophyte Index (WMI). These results have important implications for designing macrophyte surveys to track changes in biodiversity and wetland quality.

Introduction

Coastal wetlands play an important role in the biological, chemical and physical cycles of aquatic ecosystems(Carpenter 1981). They are situated at the ecotone between the terrestrial and aquatic environments and thus provide important habitat for both aquatic and terrestrial organisms (Mitsch and Gosselink 2000). Wetlands provide habitat and a source of food for fish (Jude and Pappas 1992), birds, amphibians and reptiles and they act as staging grounds for migratory birds (Maynard and Wilcox 1997, Chow-Fraser and Albert 1999).

Many of the important ecosystem services that wetlands provide are accomplished by the macrophytes (Cronk and Fennessy 2001). Macrophytes act as filters, trapping sediment and nutrients from terrestrial runoff, preventing it from reaching open water. They play a vital role in the biogeochemical cycling of nutrients in wetlands where they acts as both nutrient sinks and nutrient pumps (Cronk and Fennessy 2001). The macrophytes help reduce shoreline damage from wind and wave action by stabilizing the sediment with their root structure. The physical structure that aquatic macrophytes offer provides essential habitat for fish and macroinvertebrates, and macrophyte species richness can be directly related to the species diversity of upper trophic levels (Cronk and Fennessy 2001).

Coastal wetlands face many threats from human development, and since the arrival of European settlers we have either directly or indirectly caused the destruction of 60-80% of coastal wetlands in the Great Lakes (Smith *et al.* 1991, Ball *et al.* 2003). Coastal wetlands are generally found in areas protected from wind and waves of the open lake, and for these same reasons, they are desirable for human development. Wetlands are filled in for condo and cottage development, dredged for marinas and channels and managed for recreational use (beaches and shallow areas are mowed or raked to remove aquatic vegetation). Because of the ongoing threats that wetlands face, it is important to have the appropriate tools to facilitate the protection and conservation of our remaining coastal wetlands. One of the most valuable tools in our arsenal for wetland protection is knowledge about the species and the habitat they provide. The presence of rare or unique species in a wetland can highlight the need for its protection. In many cases the presence of rare species can be used by management agencies to protect habitat and prevent development

Currently very little research has been conducted to compare the effectiveness of different methods to sample wetland macrophytes in coastal wetlands. Methods generally vary, depending on the research goals and scope of the project. Some of the factors that researchers must consider when determining the appropriate method include: cost, time, available equipment (e.g. hip waders, canoe, small boat), number of personnel and the degree of expertise of the personnel (Hoel 1943). With the recent focus on accurate assessment of wetland biodiversity, it is also important to develop protocols that can locate rare species.

In ecology, random sampling is considered to be the gold standard (Rathburn and Gerritsen 2001), and is highly desirable when comparing the relative distribution of species within a given area or between different areas, or when making inferences about the whole population based on a subset of that population. It may not be appropriate when the goal is habitat conservation, because in many cases, habitat conservation is fueled by the desire to protect an area for its uniqueness, such as the presence of rare species that require specific niches. While the strength of random sampling is in the ability to identify the average or normal conditions, it is generally weak in identifying rare taxa, unless the site is sampled exhaustively. Since plants are generally found in clumped distributions, because of some underlying gradient such as depth or exposure, a sampling program that assumes random or even distribution of organisms may underestimate total species richness. A suitable sampling design for biodiversity assessment should therefore match the research question, and produce results that maximize the ecological rather than statistical relevance (Eberhardt and Thomas 1991).

In this study, we consider two sampling techniques that have been reported in the literature, the *grid* method and the *transect* method. The grid method requires the researcher to set up an appropriate grid pattern using poles or flagging tape throughout the entire wetland of interest, and then sample quadrats at each line intersection of the grid (e.g. Knapton and Petrie 1999). This method is associated with the most comprehensive spatial coverage of the site, and is assumed to yield the most complete species list. Although a grid pattern is relatively easy to set up in the terrestrial portion of wetlands (e.g. emergent or wet meadow zone), it is very difficult to do so in the aquatic portion of wetlands, where the water is too deep for poles to be inserted to establish the grid pattern, and where movement along straight grid lines is impossible when wading or paddling in canoes. This is generally considered the most time-consuming and labour-intensive method, and is therefore seldom used in wetland surveys.

In the transect method, researchers establish one or more (three being most common) straight lines of a standard length (e.g. 100 m) that extend from the wet meadow (terrestrial portion of the wetland) to the 1-m depth contour (the aquatic portion of the wetland) (e.g. Albert and Minc 2004?). Wetland plants are then sampled along the entire transect within a standard strip width (e.g. 1-m wide)(Cohen et al. 2004), or in a number of 1x1 m quadrats at set intervals along the transect(Bourdaghs et al. 2006). Although it is relatively easy for the researcher to identify plant species while walking within the established strip from the wet meadow to the water's edge, it is difficult to accurately identify submersed plants down to the 1-m contour unless researchers use waders or a canoe. Therefore, the transect technique is difficult to apply when sampling the aquatic portion of coastal marshes for the same reasons mentioned for the grid method above. A more serious objection is that the lower boundary of the wetland usually extends below 1 m, especially in undisturbed wetlands with good light penetration, and hence the transect method tends to underestimate the species richness of submersed aquatic vegetation, which is an important habitat component of fish (Seilheimer and Chow-Fraser 2006, Seilheimer and Chow-Fraser 2007) and benthic invertebrates (Kostuk and Chow-Fraser 2007).

In the opinion of Rathburn and Gerritsen (2001), the scientific judgement of the researcher should not be ignored, even in random sampling. They suggested a stratified random sampling approach, in which a wetland is divided into appropriate vegetation zones based on the researcher's knowledge of the ecosystem, and then randomly sampled within these zones. A stratified random sampling was also suggested to be most effective for determining tree diversity and species richness (Gimaret-Carpentier et al. 1998) when compared to random sampling in tropical forests where a strong gradient exists due to elevation. Croft and Chow-Fraser (2007) modified the stratified-random design in their *stratified* method, using their judgement to guide them to different habitat zones within the shoreline and aquatic communities of wetlands. They arbitrarily began at one habitat zone (e.g. floating vegetation) and sampled this by identifying all macrophyte species present in at least one random quadrat, and then moved to a different habitat zone (e.g. submersed aquatic bed, emergent stand, bed of floating vegetation, etc) and identified all species present in another random quadrat within that zone. This would continue until all major habitat types were sampled, and until successive transects revealed no "new" species (usually from 10-15 quadrats).

The impetus for this study was the observation that the transect method missed certain rare taxa that may be important when calculating the Wetland Macrophyte Index (WMI; Croft and Chow-Fraser 2007), a biotic index that has been used to rank the quality of fish habitat in coastal wetlands according to degree of water-quality impairment stemming from human development in watersheds and along the shoreline. The *grid* method is assumed to yield a more complete species list (including the rare species) because of the comprehensive spatial coverage; however, this method is very time consuming to conduct and as mentioned earlier, very difficult to carry out in the aquatic zone of wetlands. We hypothesized that the stratified method of Croft and Chow-Fraser (2007) would be as effective as the grid method in yielding, the total species richness, as well as the number of unique or rare species, but would be more efficient with respect to time and effort. In addition, we predict that the stratified method would yield a higher species richness, and in particular, identify more rare species, compared with the transect method. We will also determine if calculated WMI scores vary significantly among the three methods, because management of the Great Lakes coastal zone is the shared responsibility of many environmental agencies that use a variety of sampling techniques, and it is important to determine how the WMI performs with different data sources.

Study Sites

Six wetlands were chosen for this study, three pristine wetlands in Georgian Bay, and three degraded wetlands in Lake Ontario. The reason for including both pristine and degraded wetlands is to ensure that results of this study would be applicable across the degradation gradient, since we wanted to test how sampling methods would affect the performance of the WMI. The three pristine wetlands, Black Rock, Coffin Rock and Thunder Bay are located in Tadenac Bay, which has been privately owned by a fishing and hunting club for over 100 years. There is no public access permitted in Tadenac Bay and the Club limits their numbers to a maximum of 10 members at the lodge per week. As a result, Tadenac Bay has some of the most pristine wetlands in all the Great Lakes and is considered a reference site. The three wetlands we studied were embayment wetland which had low nutrients and turbidity and a diverse community of submergent, emergent, floating and meadow plants.

The three degraded wetlands are all located in the lower lakes, and included Cootes Paradise marsh, Bronte Creek and Jordan Harbour. Cootes Paradise is a large (250 ha) drowned river-mouth wetland located at the western-most end of Lake Ontario. A Remedial Action Plan (RAP) was implemented in 1992 due to high nutrients and turbidity from the Dundas Sewage Treatment Plant, urban run-off, and the feeding and spawning activity of the benthivorous common carp (*Cyprinus carpio*) (Lougheed et al. 1998). The two main restoration projects of the RAP consisted of marsh revegetation which began in 1994 followed by the exclusion of common carp in 1997 (Lougheed et al. 2004). According to Chow-Fraser's (2006) Water Quality Index (WQI), the water quality within the marsh has improved from the "highly degraded" state in 1993, to the "very degraded" category between 1994 and 1998, and by 2002, has almost reached the "moderately degraded" category. Despite the improvements in water quality, the recovery of the wetland vegetation has been slow to respond (Croft and Chow-Fraser 2007). Because of the large size of Cootes Paradise, only the northern embayment known as Hopkins Bay was used in this study.

The other two wetlands included Bronte Creek and Jordan Harbour. Bronte Creek is a riverine wetland located on the north shore of Lake Ontario in the City of Oakville. Bronte creek drains a largely urban catchment and the wetland is surrounded by houses and apartment buildings. This wetland is characterized by high turbidity levels and few submergent plants. The Jordan Harbour wetland is also a riverine wetland located on the southern shore of Lake Ontario. It is impacted by agricultural run-off from farms, vineyards and orchards in the area and is characterized by very dense submergent and floating plant growth.

Methods

Wetland surveys

Wetland macrophytes were sampled in the six wetlands with the three methods under consideration. For convenience, we will use the abbreviations 'GR', 'TR' and 'ST' when referring to the grid, transect and stratified methods. All sampling was conducted during the height of the growing season (late June to early September) 2006. Wetlands were surveyed by canoe in the deeper areas (from 0.25 to 2m) and by hip waders for the shallower areas and the wet meadow. Macrophyte presence/absence data were collected within 0.75 x 0.75m quadrats at each sampling point. All macrophytes were identified to species according to Crow and Helquist (2000) and Chaade (2002). Specimens of plants that could not be identified in the field were collected, dried and pressed if necessary, and examined more thoroughly in the lab before they could be identified. In the deeper areas, a rake was used to collect rooted plant specimens for easier identification. The boundary of the wetland was determined before sampling began so as to ensure that the same area was equally covered with all three methods. At the time of sampling, water samples were collected to determine water turbidity with a LaMotte™ portable turbidimeter.

For the GR method, quadrats were spaced roughly 10 to 15m apart in a grid pattern oriented along the longest axis of the wetland/embayment. The grid was set up with 3m long metal poles. For the TR method, transect were established along a depth gradient, in order to encounter as many habitat zones as possible. At least three transects were conducted per wetland and quadrats were spaced 5m – 10m apart along each transect line. For the ST method, a minimum of 10 quadrats were surveyed in each wetland/embayment. This method was pioneered by Croft and Chow-Fraser (2007) and makes use of the researcher's judgement to choose random quadrats within different habitat zones. For example, if the first quadrat was chosen arbitrarily in the floating zone, then quadrats located within the submergent or emergent zone would then be sampled, and so on for a minimum of 10 quadrats that covered the range of different habitat types identified visually. The spacing and distance between quadrats varied according to the heterogeneity of macrophyte zones. The sampling continued for at least ten quadrats and was considered complete if no new species were found in two consecutive quadrats. Typically, the ST method required sampling 10 to 15 quadrats. Each wetland was sampled with the three methods on the same day to reduce confounding effects of seasonal variation or differences due to meteorological conditions. The ST method was always conducted first to avoid knowledge gained from either of the other two methods to influence the selection of sites. For example, had we sampled with the GR method first and located a rare species, it would have been difficult to disregard the information when carrying out the ST method.

The time taken to sample each wetland with each method was also recorded. Because the ST method was applied first in all cases, unique or uncommon species tended to be encountered first within the ST quadrats. This inflated the time required to sample with the ST method compared with the other two, because the additional time required to key out the rare species was always attributed to the ST method. Because of this, the time taken to sample each wetland with each method had to be standardized. The corrected time accounts for this by subtracting the average time taken to sample the unique species from the total time according to the following equation:

$$\text{Corrected time} = Time - \left[\left(\frac{Time}{Total\#species} \right) * \#UniqueSpecies \right]$$

Wetland Macrophyte Index (WMI) scores

The Wetland Macrophyte Index (WMI) was developed with data obtained from coastal wetlands located in all five Great Lakes (Croft and Chow-Fraser 2007), by relating plant presence-absence data to measured water quality conditions in 127 coastal wetlands (154 wetland years). It was then validated with data from wetlands in Lakes Huron Ontario and Erie, and was proven to be a robust method for determining wetland quality. The Adjusted WMI (WMIadj), used to account for presence of exotic species, provided an index of the ecological health of the wetland ecosystem, in addition to the degree of water-quality impairment (Croft and Chow-Fraser 2007).

Geographic data

Each quadrat sampled was georeferenced with a Garmin™ Etrex GPS (4-6 m accuracy) and latitude and longitude values were imported into a GIS with ArcMap 8.2

(ESRI copyright 2002). The depth was also recorded at each quadrat with a metre stick or a weighted line marked in 10-cm depth increments.

Randomized re-sampling

To determine the relationship between species richness and sampling effort (e.g. total number of quadrats sampled and total amount of time spent sampling), we first carried out a post-hoc randomized re-sampling of the corresponding quadrats for each method in a given wetland. The first step in this procedure was to generate a randomized series of quadrats associated with the GR data by randomly selecting (random number table) 1, 3, 6, 12, 20 and 36 quadrats (with replacement). This was meant to simulate results we would have obtained had we performed a randomized sampling. We then determined the number of species that corresponded to the re-sampled quadrats (i.e. the series of 1, 3, 6, 12, 20 and 36 quadrats). For the ST and TR methods, fewer quadrats were re-sampled because fewer quadrats had been sampled originally. We also plotted species richness against the estimated time (corrected for time spent identifying unique species) that had been required to sample the quadrat series for each method in each wetland. Species richness values were transformed (squared) to produce a linear relationship for analysis of covariance.

Statistical analysis

All statistical analyses were conducted with SAS JMP IN 5.1 (SAS Institute, Cary, North Carolina, USA). We performed one-way, two-way and three-way analysis of variance (ANOVA), and whenever appropriate, used the Tukey-Kramer post-hoc test for pairwise comparison of means. The relationship between species richness and sampling effort were determined with regression analysis. The slopes and intercepts were compared with an analysis of covariance (ANCOVA).

Results

A total of over 500 quadrats were sampled within the 6 wetlands, but the number of quadrats sampled at each site differed according to the size of the study areas. Cootes Paradise was the largest, followed by Coffin rock, Black rock, Thunder bay, Jordan harbour and Bronte creek. Hence, the number of quadrats required by the GR method in Cootes was highest (61), and that for Bronte Creek was lowest (40) (Table 2.1). Regardless of size however, the total number of quadrats associated with the three methods differed significantly (ANOVA; $P < 0.0001$), and as expected, the GR method required the most effort (mean of 51.5± 3.23 SE) because of its comprehensive coverage, while the TR method required fewer quadrats (mean of 31.8±3.40 SE) and the ST method required the least (mean of 14.2±0.65 SE).

As indicated earlier, we deliberately chose sites that differed with respect to environmental quality so that our results would have widespread applicability. Since one of the obvious differences between degraded and pristine sites is water clarity (Chow-Fraser 2006), we measured water turbidity at each site to verify their status. Turbidity levels in the three Georgian Bay sites ranged from 0.40 to 1.54 NTU, and this confirms their status as pristine wetlands (Table 2.1). By comparison, the other three wetlands had much higher turbidity levels; the value for Cootes Paradise was 30.8 NTU, while that for Bronte Creek and Jordan Harbour were 14.4 and 8.7 NTU, respectively, thus confirming their status as degraded sites.

Species richness

The total number of plant species identified with all three methods (GR, ST and TR) ranged from site to site, with generally higher species richness associated with pristine sites (43 to 50), than with degraded sites (17 to 32) (Table 2.2). To account for this effect of wetland quality, we carried out a two-factor ANOVA, which tested the effect of sampling method, wetland quality, and the interaction between these. Both sampling method and wetland quality had a statistically significant effect on species richness ($P=0.0064$ and <0.0001, respectively), but there was no significant interaction between these factors ($P=0.7604$) (Table 2.3). The ST method identified significantly more species than did the other two (30.83 ± 3.93 compared with 23.83 ± 4.53 and 20.33 ± 3.99 for GR and TR, respectively; Figure 2.2a). We also found that regardless of methods used, mean species richness for the pristine sites was significantly higher than that for degraded sites (Figure 2.2b). In the case of GR and TR, there were twice as many species identified in pristine as in degraded sites. Lack of a significant effect between factors indicated that the effect of sampling method was not dependent on wetland quality.

Differences in total plant species richness among sites noted in Table 2.1 were mostly attributed to the much higher number of submergent taxa in pristine wetlands (18 to 26) compared to degraded wetlands (3 to 7); the number of emergent, floating and wet meadow taxa did not appear to vary as greatly across sites as did submergent taxa (Table 2.2). We wanted to determine how species richness of these various plant groups was affected by sampling method and wetland quality and carried out a 3-way ANOVA that accounted for plant group, wetland quality, and sampling method, as well as all possible interactions among these factors. Both plant group and wetland quality had a significant effect on species richness, and there was also a significant interactive effect between these ($P<0.0001$ for all sources) (Table 2.4). Consistent with results of the 2-way ANOVA, there were twice as many species in pristine sites as in degraded sites (8.38 vs. 4.13 (least squared mean values), respectively), although as noted previously, the major difference was noted for submergent taxa (Figure 2.3).

We also confirmed that significantly more species were identified through the ST method (7.71)(Least squared mean values) than through the GR (5.96) or TR (5.13) method ($P=0.0008$) when all data were examined together in the 3-way ANOVA model. However, once the data were sorted by plant group, sampling method was only significant for emergent taxa ($P=0.0209$), even though the significant effect of wetland quality was evident for both submergent ($P<0.0001$) and emergent ($P=0.0128$) taxa. For both pristine and degraded wetlands, mean emergent species richness associated with the ST method (8.0 and 6.3, respectively) were consistently higher than those for the GR (6.0 and 4.7, respectively) and TR (6.0 and 3.67, respectively). We found no significant effect of sampling method or wetland quality on species richness of wet meadow or floating taxa.

Unique species

The number of unique species (those found with only a single method) varied among the three sampling methods (Table 2.5), and was highest for the ST method (9.00± 0.816; 2-way ANOVA; $F=39.98$; $P<0.0001$), irrespective of wetland quality (mean of 8.33± 0.88 vs. 9.67± 1.45 for pristine and degraded wetlands, respectively). There were also significant differences among plant groups (Table 2.6), with the greatest number associated with the wet meadow group (1.83±0.57), followed by the submergent (1.17±0.29), emergent (0.89±0.28) and floating (0.11±0.08) (One-way ANOVA; $P=0.0101$; Tukey-Kramer post-hoc test; $P<0.05$). A three-way ANOVA was carried out to determine the effect of sampling method, plant group, and wetland quality on the number of unique species. There was a significant effect of plant group and sampling method on the number of unique species, but no significant effect of wetland quality (Table 2.7). When analyses were run separately for each plant group, we found a significant effect of sampling method on the number of unique species for emergent ($P<0.0001$) and wet meadow ($P=0.0005$) taxa. In both cases, the ST method identified significantly more unique taxa than the other two methods (Tukey-Kramer test; $P<0.05$).

Uncommon species

The presence of rare or endangered species is often used by managers to justify protecting habitat. Based on data from 62 wetlands (1099 quadrats) that were sampled in Georgian Bay in 2005 and 2006, we calculated frequency of occurrence for the 136 species that were found. We considered a species uncommon if it was found in less than 5% of the quadrats sampled. The mean number of uncommon species was significantly higher for ST (20.7), compared with the mean for TR (8.0) and GR (10.3) (ANOVA; $P< 0.0089$). Some examples of uncommon species encountered in Georgian Bay wetlands were creeping spearwort (*Ranunculus reptans*), floating heart (*Nymphoides cordata*), flat-leaved bladderwort (*Utricularia intermedia*), creeping bladderwort (*Utricularia gibba*), horned bladderwort (*Utricularia cornuta*), Canada blue-joint (*Calamagrostis canadensis*), and quillwort (*Isoetes spp*).

Randomized re-sampling

Species richness was plotted against number of quadrats sampled in Figure 2.5; the solid horizontal line in this figure indicates the total number of species found in a wetland by all three methods, while the dashed line represents 80% of the total. Non-linear Regression equations fitted through each set of method-wetland data were all highly significant, with correspondingly high r^2-values (Table 2.8). We performed an ANCOVA (with square transformed data) and found significant differences in slopes among the three methods ($P < 0.0010$). Slopes for the ST method were always steeper than those for the other two methods, indicating that more species were identified per unit time for this method. Much lower efficiency was associated with the GR and TR methods as indicated by the much lower slopes.

We carried out similar regression analyses on the relationship between species richness and amount of time spent sampling these quadrats (corrected for time taken to identify unique species; see Methods) (Figure 2.6). All regressions pertaining to the different methods in each wetland were significant (see Table 2.9). We found significant differences in slopes relating species richness to sampling time (corrected for time taken to identify unique species; see Methods) when all three methods were compared (ANCOVA; $P < 0.0075$ in all cases; Figure 2.6).

We used the slopes obtained from Table 2.8 and 2.9 to estimate the number of quadrats and the amount of time required to find 80% of the total species richness for each wetland (Table 2.10). For the ST method, it would require 8 to 16 quadrats, which is at least 7 times fewer quadrats compared with the TR and GR methods (54 to 146 and 54 to 179, respectively). Even if we exclude the unusually high numbers for Cootes Paradise (146 and 179, respectively for TR and GR), the average number of quadrats required to find 80% of the species within a wetland would be 13 for ST, 62 for GR and 66 for TR. We also estimated the average amount of time required to accomplish this, and predict that it would take on average 69 min with the ST method, more than twice as much time (145 min) with the GR method, and almost four-fold longer (224 min) with the TR method.

Wetland Macrophyte Index (WMI)

We used the species information to calculate both WMI and WMIadj scores for all sites and methods (Table 2.11). Sampling method had no significant effect on either set of scores (Kruskal Wallis, $n=6$; $P>0.05$), and this was true when all sites were combined for the analysis or when they were analyzed separately by wetland quality.

Discussion

The wetlands chosen for this study were selected because of the species diversity and water quality within the wetlands. Wetlands in Georgian Bay are characterized by low nutrients and low turbidity, which result in high species diversity (Chow-Fraser 2006). The turbidity levels in the three Georgian Bay sites ranged from 0.40 to 1.56 NTU (Table 2.1). The primary environmental factors that currently impact the aquatic plant communities in Georgian Bay are wind and wave exposure and prolonged lowering of the water levels. Conversely, wetlands in Lake Ontario and Lake Erie have much higher nutrient and turbidity levels that have been attributed to urban and agricultural run-off, and this can greatly affect the type of submergent plant communities that are present (McNair and Chow-Fraser 2003; McNair 2006; Chow-Fraser 2006; Croft and Chow-Fraser 2007). For example, the turbidity level for Cootes Paradise was very high (30.8 NTU) because of high levels of nutrients from urban sources (waste-treatment facility and urban runoff) and resuspension by wind and waves and bioturbation from common carp (*Cyprinus carpio*) (Chow-Fraser 2005). Bronte creek also had relatively high turbidity (14.43 NTU), because the wind can easily re-suspend the sediment in this shallow (mean depth of 30 cm) wetland. Jordan Harbour, by comparison, had the lowest turbidity (8.73 NTU) of Lake Ontario wetlands in this study, mainly because it is well protected, is deeper and has a well-established community of submergent vegetation that does not permit sediment to be easily re-suspended by wind and wave action. Given this heterogeneous range of environmental conditions encountered in these six wetlands, differences emerging from this study should have widespread applicability despite the small sample size.

Sampling with the ST method has been shown to produce the highest species richness compared to TR and GR. Although species richness depends on wetland quality, with there being higher species richness in the more pristine sites, the effectiveness of any given method was not dependent on the wetland quality. The fact that sampling method was only significant for the emergent group points to the advantage that samplers judgment has over more random methods. Many emergent species can be visually located from a long distance away. Although the sampler would not be able to positively identify a species from a long distance, it could lead them to investigate an area further, if it appeared to have new species. There was no apparent advantage of using the stratified method over the other methods for the submergent species because the submergent species are harder to locate from a long distance. When sampling the submergent zone the species that are found are largely limited by the path that the sampler takes (i.e. straight transect or a judgment guided route through the zone). The submergent species that can be seen from the boat is often limited by the glare off the water, so generally even in very clear water only the species directly beside the boat or within one or two meters can be seen. In very turbid wetlands the submergent species can not be seen through the water column so they must be reached with a rake and they are usually more sparsely distributed, so are difficult to locate, regardless of the sampling method used.

We were not surprised to encounter higher species diversity within the pristine wetlands compared to the degraded wetlands. Within pristine wetlands many of the species were found infrequently and may be only associated with a narrow depth range or a very specific niche (Appendix 1). Since the TR method may only pass through a specific depth zone a maximum of three times (depends on the number and placement of transects) the uncommon species are less likely to be found than with the ST method, where the researcher is actively searching for species. The same problem exists with the GR method, where there could be niches that lie in between sampling points that could be missed. Whereas, with the stratified method a certain zone or area with high species diversity may have two or three quadrats sampled within a small geographic area.

The number of unique species found with each method provides us with some valuable information about the effectiveness of the method. The ST method found a greater number of unique species compared to the other two methods, regardless of the wetland quality. More unique species were found with the stratified method because we were actively looking for species that had not yet been found. Since the samplers judgment was guiding them to the different habitat zones, such as areas with significant features (stream mouth, sandy or rocky areas, beaver lodge) and different depths, they would be more likely to find species that may be associated with any of those features.

But with the grid and transect method unique species would only be found if they happened to fall along the transect or grid lines.

The randomized re-sampling of this large data set has provided valuable information regarding sampling effort and sampling method. The greatest disparity among slopes of species richness to sampling effort existed for Cootes Paradise Marsh (see Figure 2.5e and Figure 2.6e), where slopes for the GR and TR methods were extremely low compared with that of ST. This reflects the very sparse distribution of the submergent macrophyte species. Because of the high turbidity in Cootes paradise there are very few submergent plants, so when sampling in a grid or along a transect there could be 5 or 6 quadrats in a row with no macrophytes at all. Since the degraded sites generally had fewer species, fewer quadrats had to be sampled, whereas in the pristine wetland the species composition was much higher and many of the species were found infrequently within the wetland, so more quadrats had to be sampled in order to find 80% of the species. Fewer quadrats were sampled with the stratified method but they were done in areas with high species diversity within a small geographic area. In the pristine wetlands the slopes of the lines are very steep after the first few quadrats had been sampled, this is because of the high species diversity found within the wetlands, so more species were found in each quadrat. Whereas in the degraded wetlands fewer species are found within each quadrat, so the total number of species increases more slowly with increasing number of quadrats. Considering that the ST method requires 7 times fewer quadrats to be sampled to identify 80% of the species, it has obvious advantages over the other methods.

The amount of time it takes to sample each wetland is a variable that can be very important to researchers because it would ultimately dictate the number of wetlands that could be sampled. We found that the ST method took half the amount of time compared to TR and GR. One of the reasons that the stratified method took less time was because transect lines or a grid pattern did not have to be set up before sampling began. Although more time was spent covering the wetland looking for species that had not yet been found, fewer quadrats were actually sampled so the overall time was less. Taking less time to sample is very attractive to many researchers because they can then sample more wetlands or sample other variables such as water quality or invertebrates or fish.

The scores for the WMI and WMIadj incorporate the scores for the individual plants species within the wetland. Within a wetland there may be several species that occupy the same niche and act as ecological analogues. So if one species was found with one method and the ecological analogue was found with another method then they would end up with the same WMI score. This is why species richness is generally not the best indicator of ecosystem health. You could have a wetland with high species diversity but those species could be indicative of poor water quality conditions. For instance, in Black rock, quillwort (*Isoetes spp*) was found with the stratified method and *Potamogeton epihydrus* was found with the grid method. Both have a U value of 5 and a T value of 2, where the U value indicates the tolerance to water quality degradation (1 is very tolerant and 5 is intolerant) and the T value indicates the niche breadth (1 is a broad niche and 3 is a narrow niche). So although these plants have different roles within a wetland (*Isoetes spp* is a rosette and *Potamogeton epihydrus* is a canopy forming submergent) they are indicators of similar water quality conditions. This redundancy that is found within ecosystems is important for adaptability to changing environments. Thus the WMI and WMIadj are robust measures of wetland quality and are independent of species richness.

One of the perceived disadvantages of the stratified method is the additional training required to determine where in the wetland to do the sampling. But considering the sampler must have sufficient training in order to identify the species, regardless of the sampling method used. Any researcher that has enough expertise to identify aquatic macrophytes in the field could easily learn to identify the different habitat zones that should be sampled.

Conclusions

Many factors must be considered when choosing a sampling protocol. The most important consideration is what the purpose or goal of the study is, and what the data will be used for. With this study we hope to highlight the importance of collecting data that is both statistically and ecologically relevant.

We had expected that the ST and GR would produce similar results for species richness and number of unique and uncommon species, but that the GR method would take more time. Our findings show that not only was the ST method more efficient in finding the greatest species richness in fewer quadrats and less time, but it was also more sensitive for locating uncommon and unique species.

The WMI has proven to be a valuable tool for determining wetland quality, but we wanted to test how sensitive it was to changes in sampling methodology. We found that there were no differences in the WMI scores calculated using data collected with the different methods. This highlights the robustness of the WMI because different species can be ecological analogues and indicate the same water quality conditions. The WMI is ideal for determining the quality of the wetland in relation to the water quality conditions, and it can be used to identify wetlands that have high quality fish habitat. But it does not directly convey information about rare species that would be valuable for conservation purposes.

We found that species richness is sensitive to the sampling method used, which implies that studies reporting species richness, but using different sampling methods should not be directly compared. Many wetland macrophyte indices have been developed using species richness as one of the metrics, and it should be noted that these indices should be used with caution considering the demonstrated sensitivity of species richness to sampling method.

This is obviously a preliminary study considering that we only compared the stratified method to two other methods and that they were only tested in 6 wetlands. But in spite of this, the results are compelling and suggest that the stratified method has advantages over the grid and transect. The stratified method will be advantageous to managers who are trying to protect wildlife habitat or who are trying to prevent development in wetlands. This study also has implications towards the conservation of other ecosystems, where accurately determining the biodiversity within the ecosystem may aid in it's conservation.

Acknowledgements

We would like to thank Jon Midwood, Lyndsay Smith, Ashley Whitehead and especially Maja Cvetkovic for their assistance during those long days while collecting data in the field. We are also indebted to the Georgian Bay Association (GBA) Foundation for providing funding and the use of the Bay Keeper, and to Mary Muter from GBA for her logistic support and inspiration while in the field.

References

Ball, H., Jalava, J., King, T., Maynard, L., Potter, B., and Pulfer, T. 2003. The Ontario Great Lakes Coastal Wetland Atlas: A summary of information (1983-1997).

Bourdaghs, M, Johnston, C.A., and Regal, R.R. 2006. Properties and performance of the floristic quality index in Great Lakes Coastal wetlands. *Wetlands* 26:718-735.

Carpenter, S.R. 1981. Submersed vegetation: An internal Factor in lake ecosystem succession. *The American Naturalist* 118:372-381.

Chow-Fraser, P. 2006. Development of the Wetland Water Quality Index for assessing the quality of Great Lakes coastal wetlands. Pages 137-166 *in* T. Simon, and T. P. Stewart, editors. Coastal Wetlands of the Laurentian Great Lakes: Health, Habitat and Indicators. Author House,Bloomington, IN, USA.

Chow-Fraser, P., and Albert, D.A. 1999. Biodiversity investment areas, coastal wetland ecosystems: Identification of "Eco-reaches" of Great Lakes Coastal Wetlands that have high biodiversity value. *State of the Lake Ecosystem Conference 1998.*

Cohen, M.J., Carsten, S., and Lane, C.R. 2004. Floristic quality indices for biotic assessment of depressional marsh conditions in Florida. *Ecol. App.* 14:784-794.

Croft, M.V., and Chow-Fraser, P. 2007. Development of the wetland macrophyte index to detect degree of water-quality impairment in Great Lakes coastal marshes (In Press). *J.Great Lakes Res.*

Cronk, J.K., and Fennessy, M.S. 2001. *Wetland Plants: Biology and Ecology.* CRC Press, Boca Raton.

Eberhardt, L.L, and Thomas, J.M. 1991. Designing environmental field studies. *Ecol. Monographs* 61:53-73.

Gimaret-Carpentier, C., Pelissier, R., Pascal, J-P, and Houllier, F. 1998. Sampling strategies for the assessment of tree species diversity. *J. Vegetation Sci* 9:161-172.

Hoel, P.G. 1943. The accuracy of sampling methods in Ecology. *The annals of mathematical statistics* 14:289-300.

Jude, D.J., and Pappas, J. 1992. Fish utilization of Great Lakes coastal wetlands. *J. Great Lakes Res.* 18:651-672.

Knapton, R.W, and Petrie, S.A. 1999. Changes in distribution and abundance of submerged macrophytes in the inner bay at Long Point, Lake Erie: implications for foraging waterfowl. *J. Great Lakes Res.* 25:783-798.

Kostuk, K., and Chow-Fraser, P. 2007. Use of zoobenthos to indicate human-induced disturbance and degree of exposure disturbance in Great Lakes Coastal Wetlands (In Press). *J. Great Lakes Res.*

Lougheed, V.L., Crosbie, B., and Chow-Fraser, P.. 1998. Predictions on the effect of common carp (Cyprinus carpio) exclusion on water quality, zooplankton and submergent macrophytes in a Great Lakes wetland. *Can. J. Fish. Aquat. Sci.* 55:1189-1197.

Lougheed, V.L., Theÿsmeÿer, T., Smith, T., and Chow-Fraser, P. 2004. Carp exclusion, food-web interactions, and the restoration of Cootes Paradise Marsh. *J. Great Lakes Res.* 30:44-57.

Maynard, L., and Wilcox, D. . 1997. *Background Report on Great Lakes wetlands*. US EPA and Environment Canada.
Mitsch, W.J., and Gosselink, J.G. 2000. *Wetlands* Third edition. John Wiley and Sons Inc, Danvers, MA.
Rathburn, S.L., and Gerritsen, J. 2001. Statistical issues for sampling wetlands. *in* R. B. Rader, D. P. Batzer, and S. A. Wissinger, editors. Bioassessment and management of North American Freshwater Wetlands. John Wiley and Sons, New York.
Seilheimer, T., and Chow-Fraser, P. 2007. Application of the Wetland Fish Index to Northern Great Lakes Marshes with Emphasis on Georgian Bay Coastal Wetlands (In Press). *J. of Great Lakes Research.*
Seilheimer, T., and Chow-Fraser, P. . 2006. Development and validation of the Wetland Fish Index to assess the quality of coastal wetlands in the Laurentian Great Lakes. *Can. J. Fish. Aquat. Sci.* 63:354-366.
Smith, P.G.R., Glooschenko, V., and Hagen., D.A. 1991. Coastal wetlands of the three Canadian Great Lakes: inventory, current conservation initiatives and patterns of variation *Can. J. Fish. Aquat. Sci.* 48:1581-1594.

Table 2.1: Total number of quadrats sampled with each method in the six study sites.

Wetland	Size of study area (ha)	Water turbidity	Sampling Method GR	ST	TR
Pristine sites					
Black Rock	1.87	1.54	58	15	24
Coffin Rock	1.64	0.40	49	17	45
Thunder Bay	1.52	1.07	55	13	35
Degraded sites					
Bronte Creek	0.49	14.4	40	13	24
Cootes Paradise	0.47	30.8	61	14	36
Jordan Harbour	0.37	8.7	46	13	27
Average for all sites			**51.5**	**14.2**	**31.8**

Table 2.2: Summary of total number of submergent, emergent, wet meadow and floating species recovered in each wetland by the three methods (GR=grid; ST=stratified; TR=transect), and when data from all three methods were combined (COMB). Numbers in bracket correspond to the percentage of species identified by each method relative to the total number of species recovered by all methods combined.

Macrophyte Habit	Method	Black Rock	Coffin Rock	Thunder Bay	Bronte Creek	Cootes Paradise	Jordan Harbour
All taxa	GR	35 (70.0)	33 (70.2)	32 (74.4)	10 (58.8)	12 (46.2)	21 (65.6)
	ST	42 (84.0)	41 (87.2)	32 (74.4)	17 (100.0)	25 (96.1)	28 (87.5)
	TR	29 (58.0)	30 (63.8)	27 (62.7)	8 (47.1)	10 (38.5)	18 (56.2)
	COMB	50	47	43	17	26	32
Emergent taxa	GR	7 (70.0)	5 (62.5)	6 (60.0)	5 (83.3)	6 (46.2)	3 (60.0)
	ST	9 (90.0)	7 (87.5)	8 (80.0)	6 (100.0)	8 (96.1)	5 (100.0)
	TR	5 (50.0)	5 (62.5)	7 (70.0)	4 (66.6.1)	5 (38.5)	2 (40.0)
	COMB	10	8	10	6	9	5
Floating taxa	GR	4 (80.0)	5 (100.0)	3 (75.0)	2 (100.0)	2 (66.7)	4 (80.0)
	ST	4 (80.0)	5 (100.0)	4(100.0)	2 (100.0)	3 (100.0)	5 (100.0)
	TR	5 (100.0)	2 (40.0)	3 (75.0)	2 (100.0)	1 (33.3)	4 (80.0)
	COMB	5	5	4	2	3	5
Wet meadow taxa	GR	8 (57.1)	6 (75.0)	9 (90.0)	1 (16.7)	2 (22.2)	9 (60.0)
	ST	12 (85.7)	5(62.5)	6(60.0)	6 (100.0)	9 (100.0)	12 (80.0)
	TR	4 (28.6)	5 (62.5)	4 (40.0)	1 (16.7)	1 (11.1)	7 (46.7)
	COMB	14	8	10	6	9	15
Submergent taxa	GR	16 (76.2)	17 (65.4)	14 (77.8)	2 (66.7)	2 (40.0)	5 (71.4)
	ST	17 (80.9)	24 (92.3)	14(77.8)	3 (100.0)	5 (100.0)	6 (85.7)
	TR	15 (71.4)	18 (69.2)	13 (72.2)	1 (33.3)	3 (60.0)	5 (71.4)
	COMB	21	26	18	3	5	7

Table 2.3: Summary of 2-way ANOVA testing the effect of sampling method, wetland quality and interaction between sampling method and wetland quality on species richness of macrophytes in wetlands.

Source	DF	SS	MS	F	p
Model	5	1638.677	327.73	15.165	0.0001
Sampling method	2	343.000	171.50	7.936	0.0064
Wetland quality	1	1283.556	1283.56	59.393	0.0001
Sampling Method * Wetland Quality	2	12.111	6.06	0.280	0.7604
Error	12	259.333	21.611		
Total	17	1898.000			

Table 2.4: Summary of 3-way ANOVA testing the effect of sampling method, wetland quality, plant group, and all possible interactions among these factors on species richness of macrophytes in wetlands.

Source	DF	SS	MS	F	P
Model	23	1313.986	57.129	11.331	0.0001
Sampling method	2	83.444	41.722	8.275	0.0008
Wetland quality	1	325.125	325.125	64.487	0.0001
Plant Group	3	411.931	137.310	27.235	0.0001
Sampling Method * Wetland Quality	2	3.000	1.500	0.297	0.7440
Plant group* Wetland quality	3	448.931	149.644	29.681	0.0001
Plant group* Sampling Method	6	23.444	3.907	0.775	0.5940
Plant group * Wetland Quality * Sampling Method	6	18.111	3.018	0.598	0.7290
Error	48	242.000	5.042		
Total	71	1555.986			

Table 2.5: Comparison of the number of species unique to a single method on a site-by-site basis.

	Sampling Method		
Wetland	**GR**	**ST**	**TR**
Pristine sites			
Black Rock	6	10	1
Coffin Rock	1	8	2
Thunder Bay	4	7	3
Degraded sites			
Bronte Creek	0	7	0
Cootes Paradise	0	12	0
Jordan Harbour	1	10	1

Table 2.6: Number of unique species grouped by plant groups: submergent (SUB), emergent (EM), floating (FL), and wet meadow (WM).

		Macrophyte Group			
Wetland	**Sampling Method**	**SUB**	**EM**	**FL**	**WM**
Pristine sites	**GR**	4	1	0	1
	ST	1	3	0	6
	TR	1	0	0	0
Black Rock					
	GR	0	0	0	1
Coffin Rock	**ST**	4	3	0	1
	TR	1	0	0	1
	GR	1	0	0	3
Thunder Bay	**ST**	2	3	1	1
	TR	2	1	0	0
Degraded sites	**GR**	0	0	0	0
Bronte Creek	**ST**	1	1	0	5
	TR	0	0	0	0
	GR	0	0	0	0
Cootes Paradise	**ST**	2	2	1	7
	TR	0	0	0	0
	GR	0	0	0	1
Jordan Harbour	**ST**	1	2	1	6
	TR	1	0	0	0

Table 2.7: Summary of 3-way ANOVA testing the effect of sampling method, wetland quality, plant group, and all possible interactions among these factors on the number of unique species in wetlands.

Source	DF	SS	MS	F	P
Model	23	138.667	6.029	7.001	0.0001
Sampling method	2	53.083	26.541	30.822	0.0001
Wetland quality	1	2.000	2.000	3.323	0.1340
Plant Group	3	27.444	9.148	10.624	0.0001
Sampling Method * Wetland Quality	2	3.583	1.791	2.081	0.1360
Plant group* Wetland quality	3	8.111	2.704	3.139	0.0337
Plant group* Sampling Method	6	28.472	4.745	5.511	0.0002
Plant group * Wetland Quality * Sampling Method	6	15.972	2.662	3.091	0.0122
Error	48	41.333	0.861		
Total	71	180.000			

Table 2.8: Summary of statistics associated with regression analysis relating species richness to total quadrats y= species richness, x = total quadrats

Wetland	Method	Equation	r^2	P
Pristine	GR	$y^2 = 22.368x + 93.416$	0.902	0.0010
Black rock	ST	$y^2 = 102.907x + 19.681$	0.912	0.0114
	TR	$y^2 = 22.134x + 28.581$	0.910	0.0031
	GR	$y^2 = 22.882x - 1.571$	0.991	0.0001
Coffin rock	ST	$y^2 = 97.071x - 98.153$	0.964	0.0029
	TR	$y^2 = 19.229x + 130.973$	0.863	0.0025
	GR	$y^2 = 19.643x + 68.624$	0.952	0.0002
Thunder Bay	ST	$y^2 = 81.596x + 14.824$	0.986	0.0006
	TR	$y^2 = 19.201x + 151.424$	0.772	0.0211
Degraded	GR	$y^2 = 2.681x - 0.758$	0.971	0.0001
Bronte Creek	ST	$y^2 = 23.395x + 3.327$	0.918	0.0101
	TR	$y^2 = 2.664x + 2.033$	0.803	0.0156
	GR	$y^2 = 2.419x - 0.888$	0.955	0.0010
Cootes Paradise	ST	$y^2 = 44.002x - 80.211$	0.929	0.0082
	TR	$y^2 = 2.915x + 4.770$	0.754	0.0247
	GR	$y^2 = 11.725x + 16.666$	0.948	0.0010
Jordan Harbour	ST	$y^2 = 53.199x - 83.551$	0.830	0.0314
	TR	$y^2 = 8.642x + 47.086$	0.889	0.0048

Table 2.9: Summary of statistics associated with regression analysis relating species richness to corrected time (min) y= species richness, x = corrected time

Wetland	Method	Equation	r^2	P
Pristine	GR	$y^2 = 14.247x + 93.416$	0.902	0.0010
Black rock	ST	$y^2 = 16.884x + 19.681$	0.912	0.0114
	TR	$y^2 = 4.427x + 28.581$	0.910	0.0031
	GR	$y^2 = 7.704x - 1.571$	0.991	0.0001
Coffin rock	ST	$y^2 = 17.089x - 98.153$	0.964	0.0029
	TR	$y^2 = 7.722x + 130.973$	0.863	0.0025
	GR	$y^2 = 9.871x + 68.624$	0.952	0.0002
Thunder Bay	ST	$y^2 = 15.083x + 14.824$	0.986	0.0006
	TR	$y^2 = 6.295x + 151.424$	0.772	0.0211
Degraded	GR	$y^2 = 2.144x - 0.758$	0.971	0.0001
Bronte Creek	ST	$y^2 = 7.957x + 3.327$	0.918	0.0101
	TR	$y^2 = 1.421x + 2.033$	0.803	0.0156
	GR	$y^2 = 1.475x - 0.888$	0.955	0.0010
Cootes Paradise	ST	$y^2 = 9.865x - 80.211$	0.929	0.0082
	TR	$y^2 = 1.503x + 4.770$	0.754	0.0247
	GR	$y^2 = 7.564x + 16.666$	0.948	0.0010
Jordan Harboür	ST	$y^2 = 8.970x - 83.551$	0.830	0.0314
	TR	$y^2 = 2.462x + 47.086$	0.889	0.0048

Table 2.10: Number of quadrats required by each sampling method to survey 80% of the total species richness for each site, and the associated time required to conduct surveys. Values calculated using equations in Table 2.9 and Table 2.10. Average values at the bottom of the table are for the number of quadrats excluding data for the GR and TR from Cootes Paradise; numbers in brackets are averages calculated without excluding Cootes data.

Wetland	Number of Quadrats			Amount of time (minutes)		
	GR	ST	TR	GR	ST	TR
Pristine sites						
Black Rock	67	15	71	105.7	93.6	355.0
Coffin Rock	62	16	67	86.6	22.8	128.7
Thunder Bay	57	14	54	183.7	88.5	166.1
Degraded sites						
Bronte Creek	69	8	69	293.9	52.0	284.7
Cootes Paradise	179	12	146	84.4	82.4	247.1
Jordan Harbour	54	14	70	112.9	77.5	163.9
Average	**62 (81)**	**13**	**66 (80)**	**144.6**	**69.4**	**224.2**

Table 2.11: Comparison of WMI and WMIadj scores (calculated according to Croft and Chow-Fraser 2007) for data obtained from the three sampling methods in each wetland in this study.

	All		GR		ST		TR	
Wetland	**WMI**	**WMI adj**	**WMI**	**WMI adj**	**WMI**	**WMI adj**	**WMI**	**WMI adj**
Black Rock	3.70	3.70	3.70	3.70	3.76	3.76	3.65	3.65
Coffin Rock	4.03	3.85	4.01	4.01	3.96	3.78	4.09	4.09
Thunder Bay	3.84	3.64	3.85	3.63	3.92	3.92	3.78	3.78
Bronte Creek	1.56	1.12	1.45	0.95	1.67	1.22	1.66	1.66
Cootes Paradise	1.34	0.99	1.28	1.28	1.45	1.10	1.35	1.35
Jordan Harbour	1.79	1.47	1.77	1.39	1.78	1.44	1.30	0.92

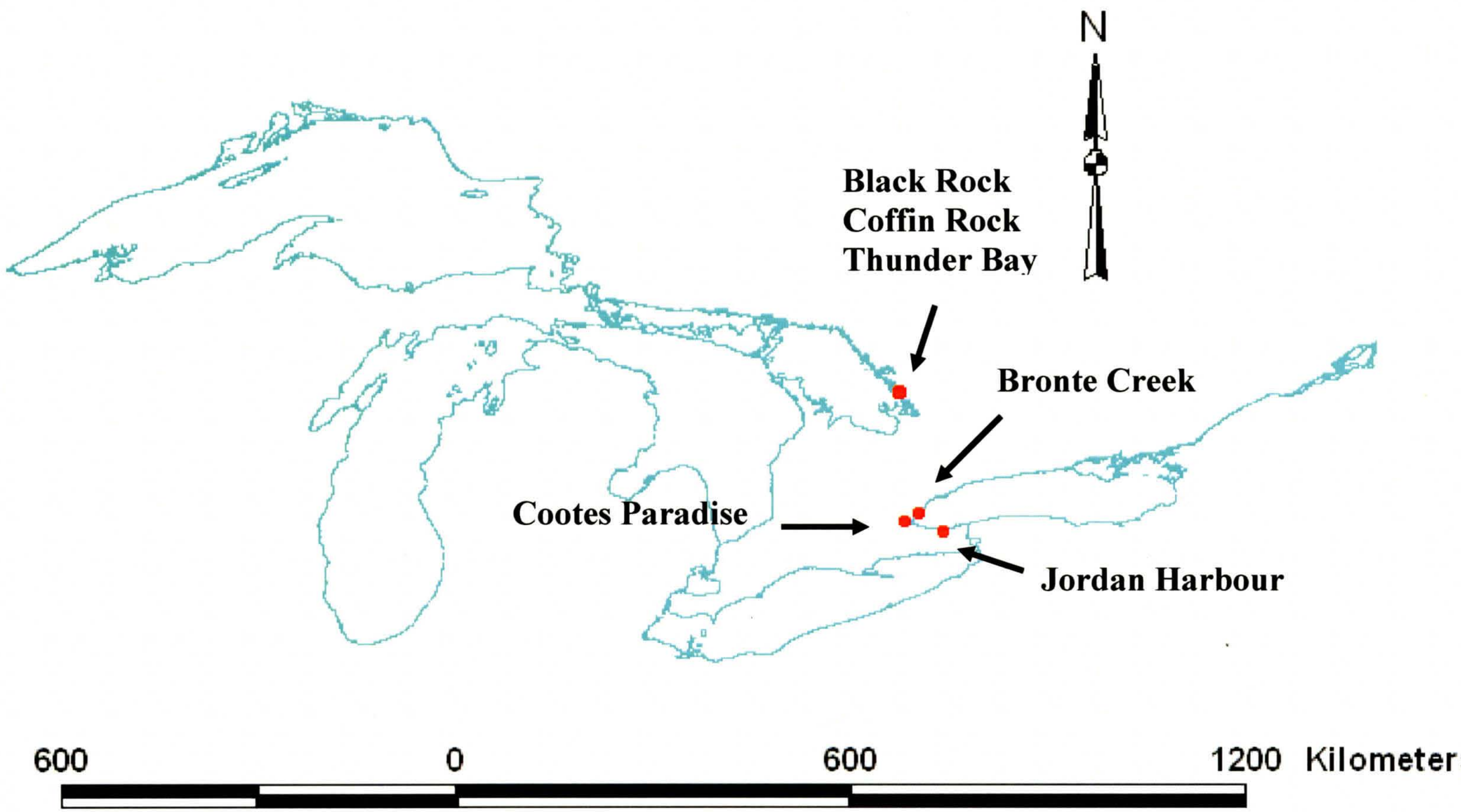


Figure 2.1: Map of the Laurentian Great Lakes showing the location of the six study sites, three in Eastern Georgian Bay, and three in Western Lake Ontario

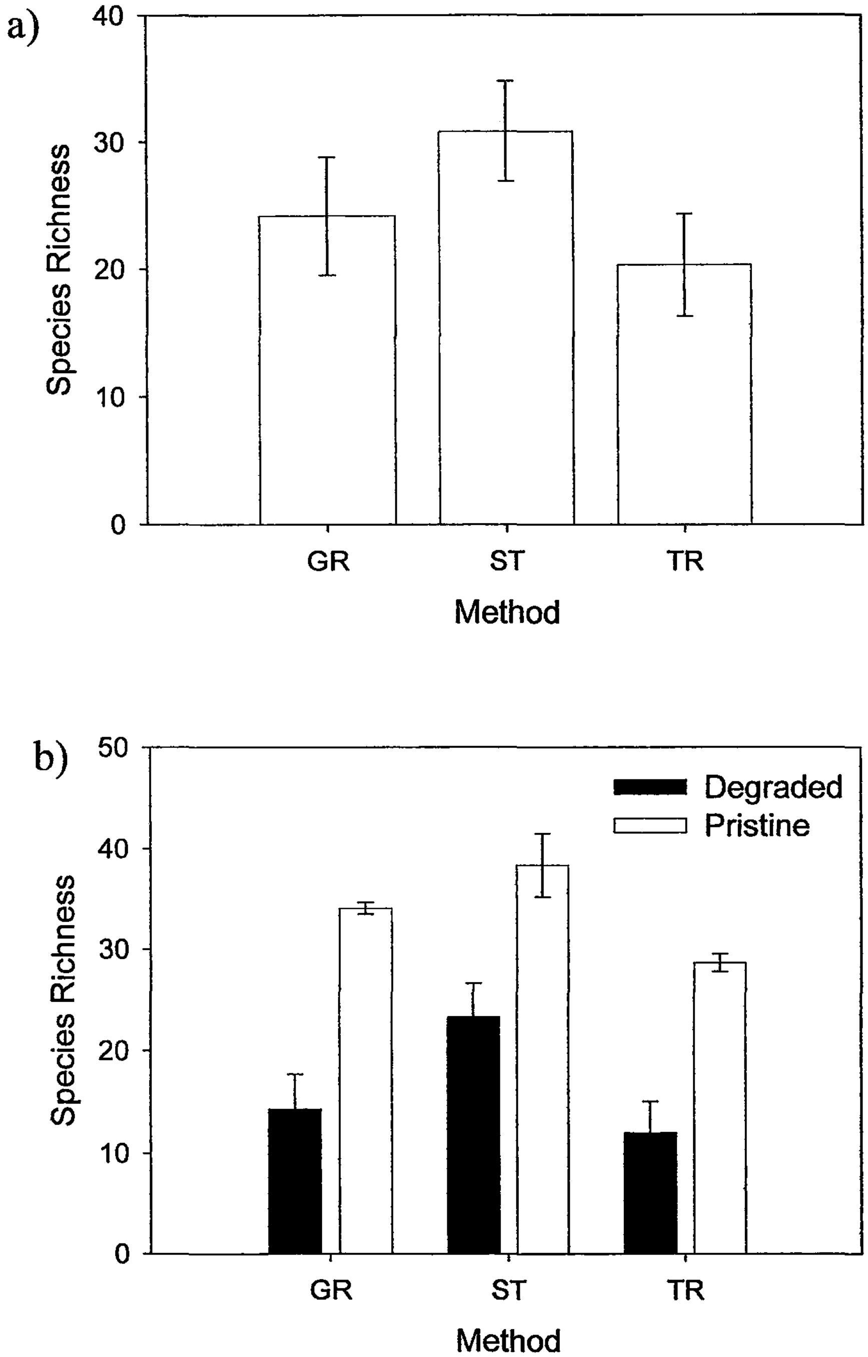


Figure 2.2: Comparison of mean species richness (± SE) associated with three sampling methods for a) all six sites in this study and b) when sites are grouped according to environmental quality (open = pristine sites; solid = degraded sites).

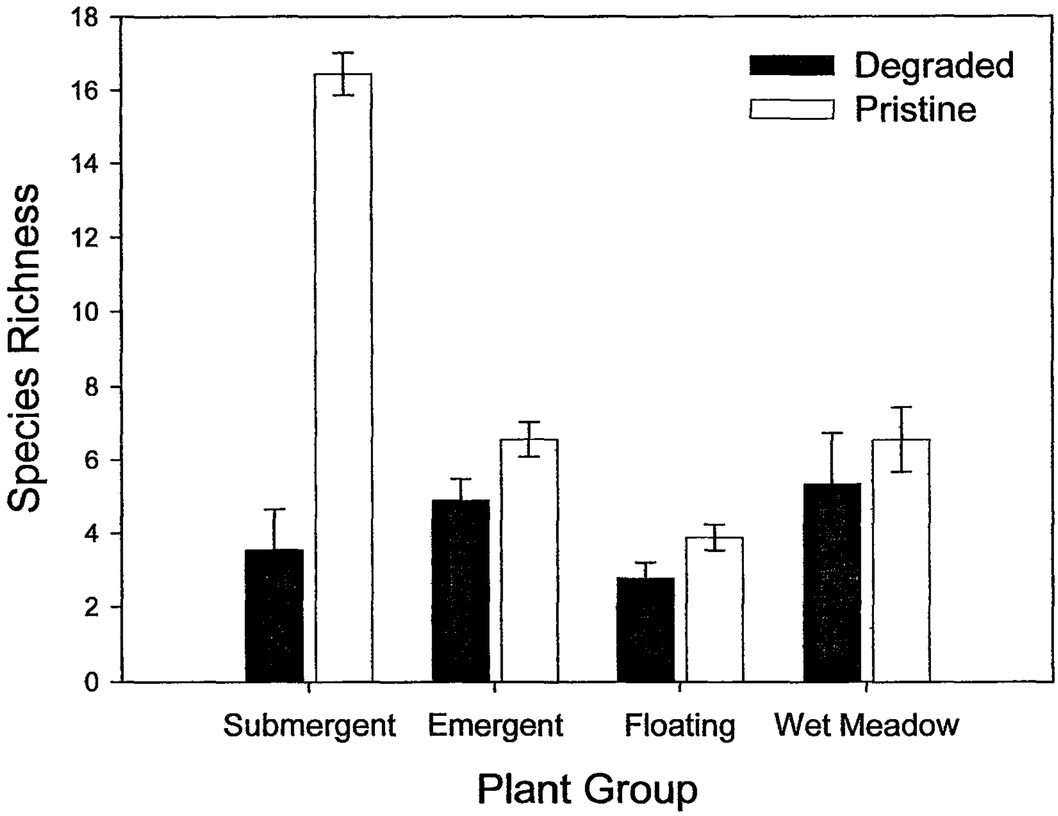


Figure 2.3: Comparison of mean species richness (± SE) associated with four plant groups sampled in pristine (open bars) and degraded (solid bars) sites.

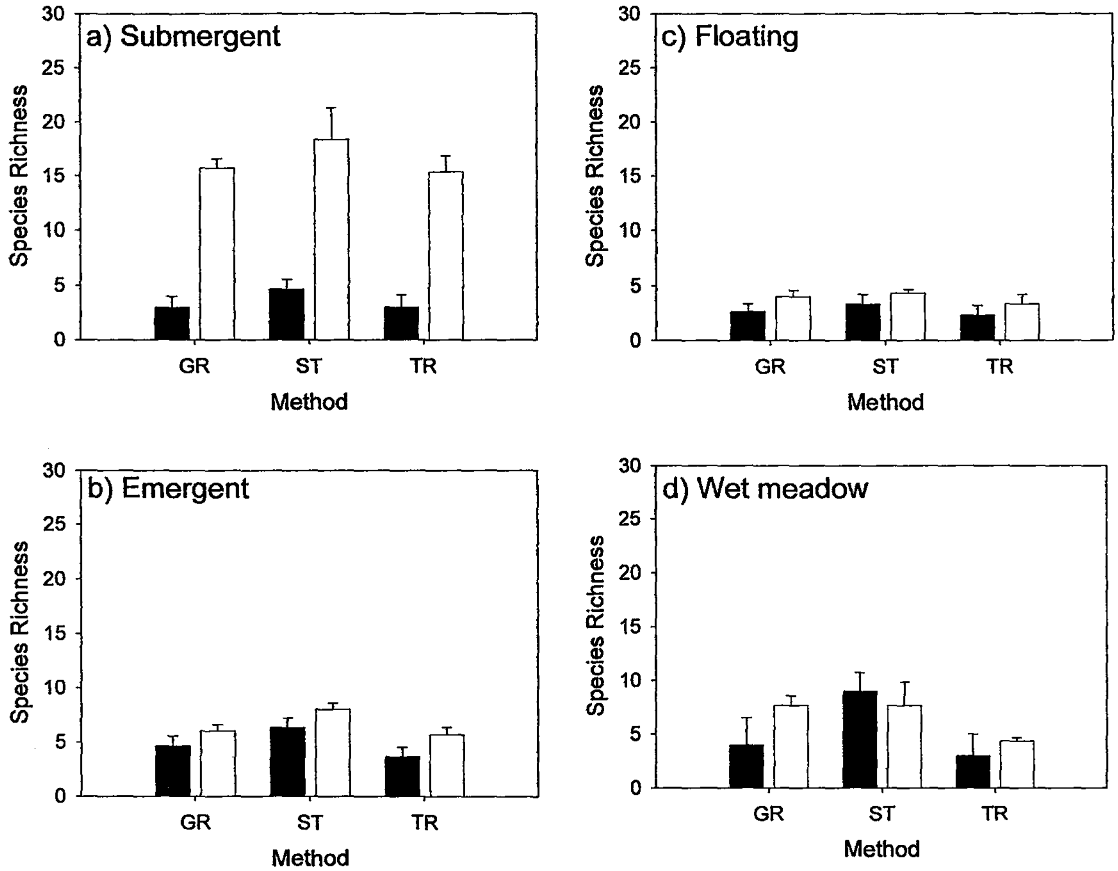


Figure 2.4: Comparison of mean species richness (± SE) associated with three sampling methods for pristine (open bars) and degraded (solid bars) sites. Data are presented separately for a) submergent taxa b) emergent taxa c) floating taxa and d) wet meadow taxa.

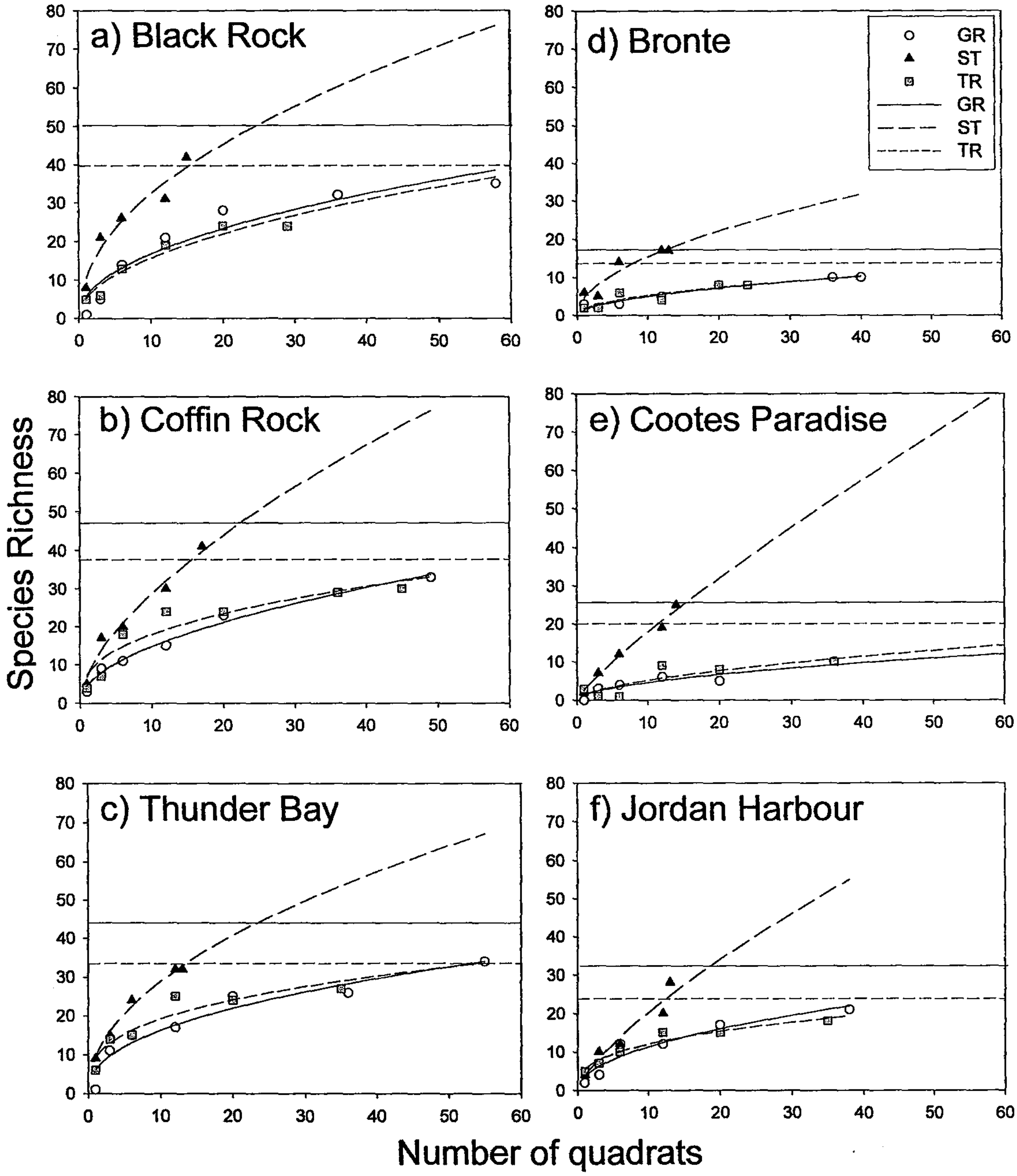


Figure 2.5: Cumulative number of species identified by each method for a) Black Rock b) Coffin Rock c) Thunder Bay d) Bronte Creek e) Cootes Paradise Marsh and f) Jordan Harbour. Horizontal lines indicate 100% (solid line) and 80% of species (dashed line) identified by all three methods. See Table 2.8 for corresponding r^2-values of method-specific regressions.

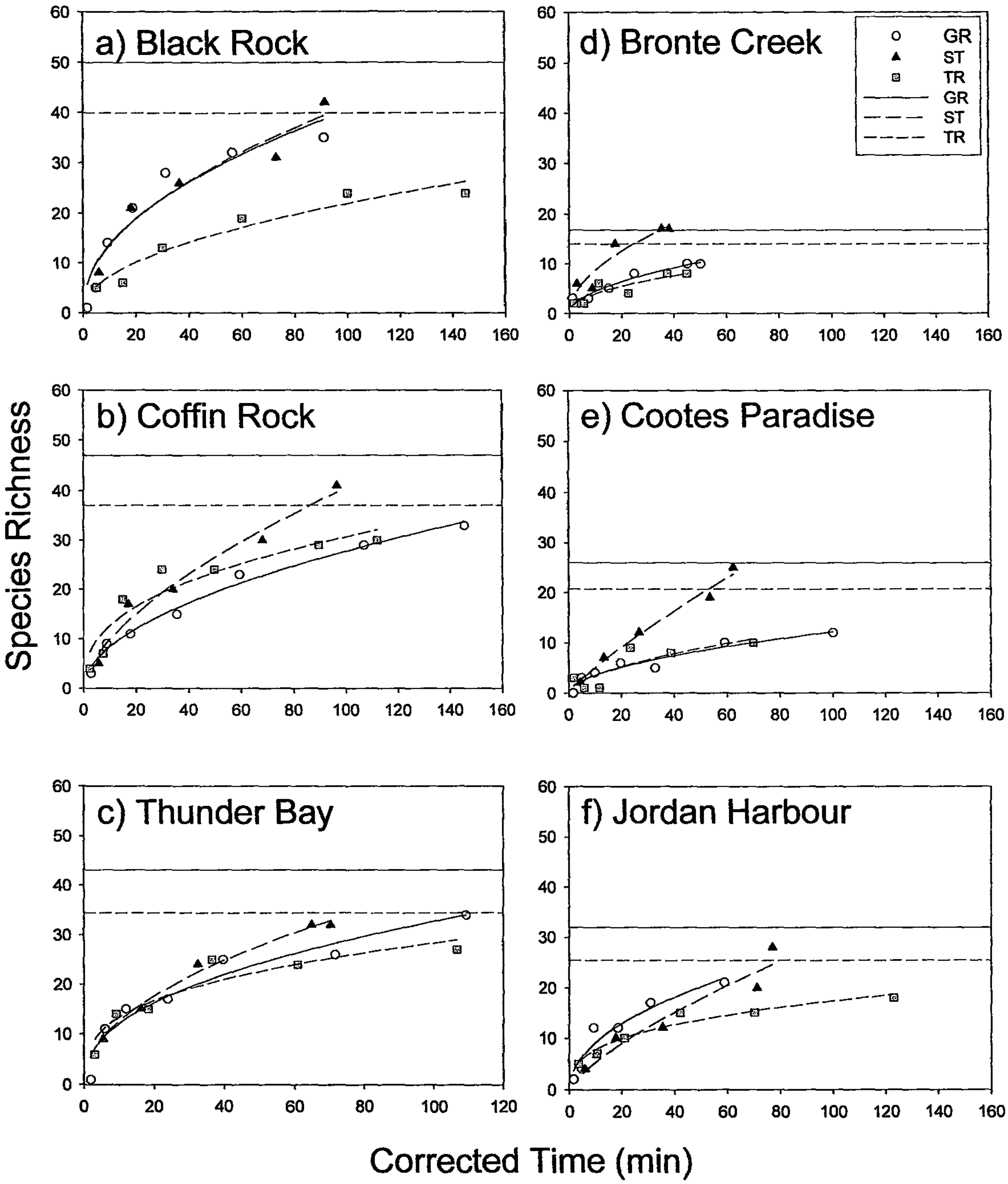


Figure 2.6: Amount of time required to complete macrophyte surveys using the three sampling methods for a) Black Rock, b) Coffin Rock, c) Thunder Bay, d) Bronte Creek, e) Cootes Paradise Marsh and f) Jordan Harbour. The time taken to identify species that were unique to only one method was subtracted from the total time. See Table 2.9 for corresponding r^2-values of method-specific regressions.

Appendix 2.1: Summary of depth measurements (cm) for species found in all 6 wetlands. Species have been divided into groups according to morphology. N indicates the number quadrats in which species were found.

Taxon	Common name	N	Mean	Median	Min	Max	Range
Porifera							
Freshwater sponges	sponges	17	49.41	46	3	89	86
Emergent taxa							
Eleocharis acicularis	needle spike rush	2	32.50	32.5	20	45	25
Eleocharis robbinsii	Robbins' spike-rush	8	17.25	18	3	26	23
Eleocharis smallii	marsh spikerush	45	14.80	14	0	62	62
Polygonum amphibium	water smartweed	5	9.80	5	1	30	29
Polygonum lapathifolium	dock-leaved smartweed	2	4.00	4	3	5	2
Polygonum sp.	smartweed	1	25.00	25	25	25	0
Pontederia cordata	pickerel weed	40	18.60	18	2	59	57
Ranunculus reptans	creeping spearwort	2	1.00	1	1	1	0
Sagittaria latifolia	broad arrowhead	18	17.11	16.5	2	33	31
Sagittaria sp.	arrowhead species	1	38.00	38	38	38	0
Scirpus acutus	hardstem bulrush	47	23.04	23	0	68	68
Scirpus americana	three-square bulrush	5	22.20	26	5	30	25
Scirpus validus	softstem bulrush	16	10.31	7.5	0	21	21
Sparganium androcladum	branched burreed	16	28.94	27	3	59	56
Sparganium eurycarpum	giant burreed	9	14.00	5	0	50	50
Sparganium sp.	burreed	3	4.00	1	0	11	11
Typha angustifolia	narrow-leaf cattail	29	16.62	15	0	60	60
Typha latifolia	broadleaf cattail	7	20.86	20	10	29	19
Typha sp.	cattail	1	3.00	3	3	3	0
Typha xglauca	hybrid cattail	42	26.55	28	0	50	50
Utricularia cornuta	horned bladderwort	1	1.00	1	1	1	0
Zizania aquatica	annual wild rice	2	25.50	25.5	23	28	5
Zizania palustris.	wild rice	79	35.19	33	1	91	90
Free-Floating							
Azolla caroliniana	mosquito fern	13	46.15	49	10	63	53
Lemna minor	lesser duckweed	157	36.66	30	0	100	100
Ricciocarpus natans	purple fringed liverwort	2	22.00	22	19	25	6
Floating rooted							
Brasenia schreberi	water shield	15	47.13	46	16	88	72

Nuphar variegata	common yellow pond lily	37	23.24	10	0	82	82
Nymphaea odorata	fragrant water lily (white)	182	33.72	29	1	92	91
Nymphoides cordata	little floating hearts	12	65.25	68	30	89	59
Potamogeton natans	broad-leaved pondweed	8	49.50	46	11	91	80
Sparganium fluctuans	floating burreed	15	53.60	58	34	69	35
Macroalgae							
Chlorophyta	filamentous algae	53	45.91	45	15	88	73
Chara sp.	muskgrass	31	48.61	46	3	91	88
Nitella sp.	stonewort	4	44.75	32	24	91	67
Meadow							
Amblystegiaceae family	moss	1	5.00	5	5	5	0
Asclepias incarnata	swamp milkweed	1	0.00	0	0	0	0
Calamagrostis canadensis	Canada blue joint	6	0.00	0	0	0	0
Carex sp.	sedge	97	4.58	0	0	45	45
Cirsium sp.	thistle	2	0.00	0	0	0	0
Cuscuta gronovii	swamp dodder	3	0.00	0	0	0	0
Dulichium arundinaceum	three-way sedge	56	9.20	8	0	28	28
Epilobium ciliatum	northern willow herb	1	28.00	28	28	28	0
Eupatorium perfoliatum	boneset	6	0.33	0	0	1	1
Glyceria grandis	American manna grass	10	5.00	1	0	25	25
Impatiens capensis	spotted jewelweed	18	6.22	0	0	45	45
Iris versicolor	blue-flag iris	6	1.50	0	0	8	8
Juncus effusus	soft rush	37	2.57	1	0	20	20
Juncus sp.	rush	48	14.92	9	0	77	77
Lobelia cardinalis	cardinal flower	1	0.00	0	0	0	0
Lycopus uniflorus	northern water horehound	2	9.00	9	0	18	18
Lythrum salicaria	purple loosestrife	3	3.33	5	0	5	5
Mimulus ringens	square stemmed monkey flower	5	19.00	3	0	63	63
Myrica gale	sweet gale	3	0.00	0	0	0	0
Onoclea sensibilis	sensitive fern	5	0.00	0	0	0	0
Rorippa palustris	marsh yellow cress	1	18.00	18	18	18	0
Rubus idaeus	raspberry	1	0.00	0	0	0	0
Salix sp.	willow	1	0.00	0	0	0	0
Solanum dulcamara	climbing night shade	2	1.50	1.5	0	3	3
Urtica dioica	stinging nettle	5	9.00	0	0	45	45
Verbena hasta	blue vervain	6	4.83	0	0	29	29

Submergent unrooted							
Ceratophyllum demersum	coontail	29	56.03	55	28	88	60
Utricularia gibba	creeping bladderwort	12	22.92	19	3	59	56
Utricularia intermedia	flatleaved bladderwort	11	28.73	21	2	91	89
Utricularia purpurea	purple bladderwort	9	51.89	47	34	81	47
Utricularia sp.	bladderwort	1	38.00	38	38	38	0
Utricularia vulgaris	common bladderwort	10	43.20	44.5	21	60	39
Submergent rooted							
Bidens beckii	water marigold	5	50.60	40	30	86	56
Callitriche sp.	water starwort	7	12.43	9	1	33	32
Elodea canadensis	Canadian waterweed	19	42.05	38	13	88	75
Eriocaulon aquaticum	pipewort	44	26.82	25.5	0	70	70
Hippurus vulgaris	mare's tail	4	31.00	34	18	38	20
Isoetes sp.	quillwort	1	40.00	40	40	40	0
Myriophyllum heterophyllum	two-leaf water milfoil	2	29.00	29	13	45	32
Myriophyllum sibiricum	common water milfoil	7	36.71	38	10	58	48
Myriophyllum spicatum	Eurasian water milfoil	57	60.58	59	28	100	72
Myriophyllum tenellum	slender water milfoil	6	69.50	76.5	21	89	68
Najas flexilis	slender water nymph	60	50.87	47.5	13	91	78
Potamogeton amplifolius	large-leaved pondweed	4	88.25	86	76	105	29
Potamogeton crispus	curly-leaf pondweed	3	27.67	27	26	30	4
Potamogeton epiphydrus	ribbon-leaf pondweed	2	39.50	39.5	33	46	13
Potamogeton filiformis	thread leaf pondweed	3	28.00	28	26	30	4
Potamogeton foliosus	leafy pondweed	12	31.42	29	23	55	32
Potamogeton gramineus	variable pondweed	59	41.36	38	3	89	86
Potamogeton pusillus	slender pondweed	2	48.00	48	46	50	4
Potamogeton richardsonii	clasping-leaved pondweed	2	67.00	67	46	88	42
Potamogeton robbinsii	fern-leaf pondweed	42	43.79	39.5	3	88	85
Potamogeton spirillus	northern snailseed pondweed	10	41.10	38	15	76	61
Sagittaria graminea	grassy arrowhead	71	31.76	29	0	77	77
Scirpus subterminalis	water bulrush	101	30.49	28	9	91	82
Stuckenia pectinatus	sago pondweed	4	58.50	57	39	81	42
Utricularia subulata	slender bladderwort	18	61.67	62	27	105	78
Vallisneria americana	tape grass	20	62.65	63.5	30	105	75